Changes in Exposure to $PM_{2.5}$ in English Dwellings: An Unintended Consequence of Energy Efficient Refurbishment of the Housing Stock.

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Statement of Originality

I, Clive Shrubsole, the author of this thesis, confirm that the work presented in this thesis has been composed by myself. Where the work of others has been used, including information derived from other sources, it is fully acknowledged in the text and in captions to tables and illustrations. This work has not been nor will be submitted in part, or whole for any other degree, diploma or similar qualification. In addition, this thesis does not exceed 100,000 words in length, excluding the appendices. It is confirmed that all references were correct at the time of submission.

Signed

Date 23rd October 2017

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Much of this research was not carried out in isolation, but as part of various other projects with numerous partnering organisations and individuals. My specific role within these projects and the contribution of others is clarified and acknowledged in sections 1.5-1.6 and throughout the text where appropriate.

Finally, to my partner (now fiancée), best friend and soulmate Lea; for her time when I'm sure she had better things to do, for advice, grammar checks, proof reading, suggestions, patience, encouragement, emotional support and endless cups of tea; thank you from the bottom of my heart.

"For some socio-technical systems, simulation is the only way we know of investigating their future states - If you do not trust a carefully executed simulation, you probably have less reason to trust anything else, including the way you currently make decisions."

Jeffrey Johnson

Abstract

UK legislation will result in energy efficiency gains through increased insulation, and airtightness in UK housing in the coming decades. This limited-focus policy approach has led to an array of possible unintended consequences, including likely changes in Indoor Air Quality (IAQ) and exposure profiles for airborne pollutants such as $PM_{2.5}$. Quantification of any changes in indoor concentrations of $PM_{2.5}$ is needed due to known impacts on population health. This thesis seeks to address whether the introduction of energy efficiency and ventilation strategies will lead to negative unintended consequences by increasing $PM_{2.5}$ concentrations in English dwellings, or provide health co-benefits by reducing indoor $PM_{2.5}$; what factors influence such concentrations and whether their contribution can be quantified? Its geographical focus is the English housing stock commencing with London, comparing London with another location (Milton Keynes) and finally extending to the whole English stock. It considers possible differences in exposure as experienced by different income groups and tenures. It investigates the range of interacting factors that contribute to indoor $PM_{2.5}$ exposures including for example; external meteorological conditions/pollutant concentrations; location; building characteristics; ventilation type; indoor sources; occupant income and behaviour. Such complexity requires a modelling approach. Building archetypes representative of English dwellings and validated ventilation and indoor pollutant simulation techniques are used to model both current and future changes in indoor $PM_{2.5}$ exposures.

Highlights of the research findings include (1) The application of purpose provided ventilation and removal of indoor generated $PM_{2.5}$ at source are critical to the overall reduction of indoor exposure in most cases; (2) Increasing envelope airtightness alone reduces ventilation heat loss, assisting CO_2 reduction targets whilst also reducing ingress of external $PM_{2.5}$, but substantially increases indoor sourced $PM_{2.5}$ concentrations with possible overall negative health consequences; (3) Building characteristics, location, income level and occupant behaviour influence individual exposure where energy efficiency measures are implemented; (4) Households below the low income threshold are more likely to experience greater indoor $PM_{2.5}$ concentrations, although further monitoring research is needed to confirm/refute this; (5) The models constructed for this study have a possible wider applicability for other airborne pollutants, locations, and building stocks.

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List of Abbreviations and Acronyms

AURN	*Automatic Urban and Rural Network*
EHS	*English House Survey*
FDMS	*Filter Dynamic Measurement System*
GHG	*Green House Gas*
GLA	*Greater London Area*
HIA	*Health Impact Assessment*
IAQ	*Indoor Air Quality*
LAQN	*London Air Quality Network*
LHSTM	*London School of Hygiene and Tropical Medicine*
LIT	*Lower Income Threshold*
mE	*Micro Environmental*
MVHR	*Mechanical Ventilation with Heat Recovery*
OSPM	*A practical street pollution model, developed by the Department of Environmental Science at Aarhus University*
PPV	*Purpose Provided Ventilation*
PSD	*Participatory System Dynamics*
$PM_{2.5}$	*Particulate Matter having an aerodynamic diameter equal to or less than 2.5microns*
SCRIBE	*Strategies for Carbon Reduction In the Built Environment.-A modelling tool*
TEOM	*Tapered Elemental Oscillating Microbalance*

Thesis Associated Publications

Peer Reviewed Journal Papers

(**Citation indices for papers are from Scopus, tandfonline. and Web of Science Core Collection)**

Shrubsole, C., Macmillan, A., Davies, M. and May, N. (2014). *100 Unintended consequences of policies to improve the energy efficiency of the UK housing stock*. Indoor and Built Environment, 23(3), 340-352. Doi: 10.1177/1420326X14524586.
Contributed as lead author and researcher; this paper scopes the range and domains of impacts of ventilation and energy efficiency strategies, highlighting indoor air quality as a key issue *[Chapter 2]* **Awarded Best paper 2014: Indoor and Built Environment: Sage Publishing**
Citations: 17

Shrubsole, C., Ridley, I., Biddulph, P., Milner, J., Vardoulakis, S., Ucci, M. and Davies, M. (2012). *Indoor $PM_{2.5}$ exposure in London's domestic stock: modelling current and future exposures following energy efficient refurbishment*. Atmospheric Environment, 62, 336-343.
Contributed as lead author and researcher; this paper models current and possible futures exposures to $PM_{2.5}$ under a specific refurbishment scenario using CONTAM and considers occupant behaviour impacts and differential sensitivity analysis *[**Chapters 4**]*
Citations: 29

Shrubsole, C., Das, P., Milner, J., Hamilton, I.G., Oikonomou, E., Spadaro, J.V., Davies, M. and Wilkinson, P. (2015). *A tale of two cities: comparative impacts of CO_2 reduction strategies on dwellings in London and Milton Keynes* Atmospheric Environment, 120, 100–108.
Contributed as lead author and researcher; helping to develop the SCRIBE tool, this paper models how location influences domestic CO_2 reduction strategies impacting indoor $PM_{2.5}$ exposure and health *[**Chapter 5**]*
Citations: 1

Shrubsole, C., Taylor, J., Das, P., Hamilton, I.G. and Davies, M. (2015) *Impacts of energy efficiency retrofitting measures on indoor $PM_{2.5}$ concentrations across different income groups in England: a modelling study*. Advances in Building Energy Research. doi10.1080/17512549.2015.1014844
Contributed as lead author and researcher; this paper models social effects around household income level and the impacts on indoor $PM_{2.5}$ exposures *[**Chapter 6**]*
Citations: 4

Hamilton, I.G., Milner, J., Chalabi, Z., Das, P., Jones, B., **Shrubsole, C**., Ridley, I., Davies, M. and Wilkinson, P. (2015) *Health effects of energy efficiency interventions in England: a modelling study*. BMJ Open. http://bmjopen.bmj.com/content/5/4/e007298.full.pdf+html
Contributed as author and researcher, developed modelling for the ventilation, energy efficiency and pollutant elements of this work. *[**Appendix G**]*
Citations: 9

Das, P., **Shrubsole, C**., Davies, M., Mavrogianni, A., Taylor, J., Jones, B. and Chalabi, Z. (2014). *Using probabilistic sampling-based sensitivity analyses for indoor air quality modelling.* Building and Environment, 78, 171-182. Doi: 10.1016/j.buildenv.2014.04.017
Contributed as 2nd author, assisted in developing a stock model of indoor air quality (including $PM_{2.5}$), applying a range of sensitivity analyses, selecting determinants to build a reliable metamodel. *[**Chapter 7**]*
Citations: 14

Taylor, J., **Shrubsole, C**., Davies, M., Biddulph, P., Das, P., Hamilton, I. and Oikonomou, E. (2014). *The modifying effect of the building envelope on population exposure to $PM_{2.5}$ from outdoor sources*. Indoor Air. Doi: 10.1111/ina.12116
Contributed as 2nd author, research provided variable inputs to modelling. Using EnergyPlus, this paper quantifies the impacts of the building envelop on indoor exposure to $PM_{2.5}$ from outdoor sources in London. *[**Chapter 5**]*
Citations: 14

Taylor, J., **Shrubsole, C**., Biddulph, P., Das, P., Davies, M., & Jones, B. (2014). *Simulation of pollution transport in buildings: The importance of taking into account dynamic thermal effects*. Building Services Engineering Research and Technology, 35 (6), 682-690.
Doi: 10.1177/0143624414528722
Contributed as 2nd author, provided CONTAM modelling outputs. This paper Contrasts predictions of indoor $PM_{2.5}$ concentrations using CONTAM and EnergyPlus software *[**Chapter 7**]*
Citations: 2

Taylor, J., Mavrogianni, A., Davies, M., Das, P., **Shrubsole, C**., Biddulph, P. & Oikonomou (2015). *Understanding and mitigating overheating and indoor $PM_{2.5}$ risks using coupled temperature and indoor air quality models* Building Services Engineering Research and Technology, 0 (0), 682-690. Doi: 10.1177/0143624414566474
Contributed as author, provided modelling inputs and text. This paper demonstrates the relationship between indoor $PM_{2.5}$ and temperature in housing using EnergyPlus software *[**Chapter 7**]* **Winner of the CIBSE Napier Shaw Bronze Research Medal, which is awarded for the most highly rated research paper published in BSER&T**
Citations: 9

Milner, J., **Shrubsole, C**., Das, P., Jones, B., Ridley, I., Chalabi, Z. & Wilkinson, P. (2014). *Home energy efficiency and radon related risk of lung cancer: modelling study*. BMJ (Clinical research ed.), 348. Doi: 10.1136/bmj.f7493
Contributed as 2nd author, produced CONTAM and future scenarios models. Using a different pollutant (radon) combining exposure outputs with health impact assessments, this paper explores changes in population exposure, highlighting the wider use of the modelling. *[**Appendix G**]*
Citations: 27

Jones, B., Das, P., Chalabi, Z., Davies, M., Hamilton, I. G., Lowe, R., **Shrubsole, C**. & Wilkinson, P. (2013). *The Effect of Party Wall Permeability on Estimations of Infiltration from Air Leakage*. International Journal of Ventilation, ISNN 12 (1)1473-3315.
Contributed as author, this paper considers impacts of party wall permeability on air flow and, which may impact on indoor $PM_{2.5}$ concentrations. *[**Chapters 3&7**]*
Citations: 8

Das, P., Chalabi, Z., Jones, B., Milner, J., **Shrubsole, C**., Davies, M. & Wilkinson, P. (2013). *Multi-objective methods for determining optimal ventilation rates in dwellings*. Building and Environment, 66, 72-81. Doi: 10.1016/j.buildenv.2013.03.021
Contributed as an author, provided CONTAM modelling inputs. This papers explores differences in building geometry and ventilation strategies for a range of pollutants including $PM_{2.5}$. *[**Chapter 4**]*
Citations: 18

Macmillan, A., Davies, M., **Shrubsole, C**., Luxford, N., May, N., Chiu, L-F., Trutnevyte, E., Bobrova, K., Chalabi, Z. (2016). *Integrated decision-making about housing, energy and wellbeing: a qualitative system dynamics model"* Environmental Health (special issue) Healthy-Polis: Challenges and Opportunities for Urban Environmental Health and Sustainability
Contributed as an author, provided input into the design of the study, analysis of the data and organising and running the workshop. This papers highlights the need for more integrated policy formation and application to prevent unintended consequences such as increases in indoor $PM_{2.5}$ concentrations following energy efficient refurbishment. ***[Chapters 2]*** **Most cited article 2016 Environmental Health**
Citations: 5

Nix, E., Das, P., Taylor, J., **Shrubsole, C**., Chalabi, Z., Davies, M., Mavrogianni, A., Milner, J., Wilkinson, P and Ucci, M. (Submitted) *City-wide mapping of household health-energy inequalities: Implications for housing interventions in Delhi, India*.
Contributed as an author and provided input data for modelling. ***[Appendix G]***

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Doi: 10.1016/j.buildenv.2016.01.010
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Shrubsole, C., Biddulph, P., Chalabi, Z., Davies, M., Milner, J., Ridley, I. and Wilkinson, P. *Decarbonising London's domestic stock: Implications for indoor $PM_{2.5}$ exposure and health*. Proceedings of the International Conference on Air Quality 2012 - Athens, Greece.
Contributed as lead author, researcher and speaker; this paper describes the development of models for current and possible futures exposures in London to $PM_{2.5}$ under refurbishment scenarios using CONTAM and considers occupant behaviour impacts and differential sensitivity analysis. [***Chapters 5]***

Vardoulakis, S., **Shrubsole, C**., Davies, M., Biddulph, P., Wilkinson, P., Chalabi, Z., Milner, J., Ridley, I., Lai, K. and Ucci, M. *Public health outcomes of climate change mitigation and adaptation policies in the built environment*. Proceedings of the Healthy Buildings Conference 2012 - Brisbane, Australia.
Contributed as 2nd author and researcher (pollutant modelling); this paper explores the health implications of changes in $PM_{2.5}$ exposures as an example of outcomes of mitigation policies. ***[Chapter 5]***

Shrubsole, C., Das, P., Hamilton, I. and Davies, M. *Comparative impacts of CO_2 reduction targets on indoor air quality and energy use in domestic properties in London and Milton Keynes 'a tale of two cities'.* Proceedings of the Public Health England Annual Review Meeting: Outdoor and Indoor Air Pollution Research 2014 – Solihull UK.
Contributed as lead author, researcher and speaker; this paper describes the SCRIBE tool and how location influences domestic CO_2 reduction strategies impacting indoor $PM_{2.5}$ exposure and health. ***[Chapter 5]***

Milner, J., **Shrubsole, C**., Payel, D., Chalabi, Z., Wilkinson, P., Davies, M. *Unintended consequences of climate change mitigation for radon-related lung cancer.* Proceedings of the 2013 IEH Annual UK Review Meeting on Outdoor and Indoor Air Pollution Research-Cranfield UK
Contributed as lead author, researcher and speaker; this paper describes possible changes in population exposure to radon under future building interventions for climate change objectives. ***[Appendix G]***

Taylor, J., Davies, M., **Shrubsole, C**., Das, P., Chalabi, Vardoulakis, S. *Indoor $PM_{2.5}$ exposure modelling from traffic sources using GIS and building simulation.* Proceedings of the 2013 IEH Annual UK Review Meeting on Outdoor and Indoor Air Pollution Research - Cranfield UK
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Contributed as author, developed the modelling ***[Appendix G&H]***

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Contributed as an author and carried out on-site surveying and production of archetypes for modelling. ***[Appendix G]***
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Contributed as an author and supervised the MSc that generated the study ***[Appendix G]***

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Conference Presentations to which the author contributed

Housing energy efficiency and radon-related health risks. Wilkinson, P., Milner, J., **Shrubsole, C**., Chalabi, Z. and Davies, M. ISEE Conference 2012 - Columbia, USA. ***[Appendix G]***

The relationship between permeability and infiltration in conjoined dwellings.
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The UK 2050 climate change abatement policies for housing energy efficiency: health impact and implications for policymakers. Hamilton, I. G., Milner, J., Chalabi, Z., Das, P., Jones, B., **Shrubsole, C**., Ridley, I., Davies, M. and Wilkinson, P. Environmental Health Conference 2013- Boston, USA. ***[Chapter 6]***

Do optimal rates of air exchange with the external environment in naturally-ventilated dwellings depend on built form? Das, P., **Shrubsole, C**., Chalabi, Z., Davies, M., Hamilton, I., Jones, B.M., Milner, J. and Wilkinson P. Given by PD, European Symposium on Building Physics – CESBP 2013 Vienna, Austria. ***[Chapter 6]***

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The unintended consequences of energy efficient retrofit on indoor air pollution, condensation and overheating risk in a typical Georgian mid-terrace house.
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100 unintended consequences of policies to create a low-carbon housing stock.
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Using probabilistic sampling-based sensitivity analyses for indoor air quality modelling. Das, P., **Shrubsole, C**., Davies, M., Mavrogianni, A., Taylor, J., Jones, B. and Chalabi, Z. Indoor Air Conference 2014 - Hong Kong. ***[Chapter 7]***

Understanding the health impact of future energy efficiency and low-emissions in London. Hamilton, I., Milner, J., Das, P., Chalabi, Z., **Shrubsole, C**., Wilkinson, P. and Davies. M. Urban Environmental Pollution Conference 2014 - Toronto, Canada. ***[Chapter 6]***

Modelling the modifying effect of homes on population exposure to pollution.
Taylor, J., **Shrubsole, C**., Davies, M., Wilkinson, P., Mavrogianni, A., Milner, J., Das, P. and Chalabi, Z. ISEE Conference 2014 - Seattle, Washington, USA. ***[Chapter 5]***

Health impacts and cost-effectiveness of home energy efficiency interventions in England.
Milner, J., Hamilton, I., Chalabi, Z., Das, P., Jones, B., **Shrubsole, C**., Davies, M. and Wilkinson, P. UKIEG Conference 2015 London UK. ***[Chapter 6]***

Improving the energy efficiency of the English housing stock: Implications for health and cost-effectiveness. Milner, J., Hamilton, I., Chalabi, Z., Das, P., Jones, B., **Shrubsole, C**., Davies, M. and Wilkinson, P. ISEE Conference 2015. ***[Chapter 6]***

Modelling population exposure to high indoor temperatures under changing climates, housing conditions, and urban environments in England. Taylor, J., Symonds, P., Mavrogianni, A., Davies, M., **Shrubsole, C**., Hamilton, I., Chalabi, Z., Wilkinson, P. ICUR Conference 2016 Lisbon Portugal. ***[Chapter 4]***

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Quantifying comfort and monetizing the health impact of household energy efficiency improvements: Documentation for the HIDEEMv2 (Health Impact of Domestic Energy Efficiency Measures version 2) model. For the Department of Energy and Climate Change (DECC) June 2012

Evaluation of CONTAM models used in the development of the HIDEEM. Report for the Institute of Environmental Design & Engineering: Copyright © May 2014 University College London.

Modelling of build environment exposures relevant to health for the Public health impacts in urban environments of greenhouse gas emissions reduction strategies (Purge). Project reports for Phase 3 and Final reports for WP1, WP4, Summary for Policy Makers and Dissemination Guidelines. E.U 7th Framework Programme Grant agreement no: 265325 September 2013 & 2014.

Tutorial and Design Exercises for Airflow Studies Using CONTAM 3.0.1 for MSc Environmental Design & Engineering: Copyright © June 2014 University College London.

Report on the mapping work for stakeholders. Integrated decision-making about housing, energy and wellbeing (HEW) Project. EPSRC five-year Platform Grant on Complex Built Environment Systems (CBES) March 2014.

Modelling of build environment exposures relevant to health for the Public health impacts in urban environments of greenhouse gas emissions reduction strategies (PURGE). Project reports for Phase 3 and Final reports for WP1, WP4, Summary for Policy Makers and Dissemination Guidelines. E.U 7th Framework Programme Grant agreement no: 265325 September 2013 & 2014.

Report for proposed MSc Programme. Title: Health, Wellbeing and Sustainable Buildings - 2017/18 University College London.

MSc Module: BENVGEEF Indoor Air quality in Buildings Production and delivery of module. Institute of Environmental Design and Engineering October 2015 University College London

Chapter 1

Background

Introduction

Housing in the UK is responsible for one quarter of the UK total end-user CO_2 emissions, with over half of this produced from space heating (DECC, 2014). Motivated by ambitious CO_2 reduction targets - an 80% reduction of CO_2 emissions from 1990 levels by 2050, concerns over fuel poverty, energy security/cost and in response to the EU Energy Performance of Buildings Directive (EPBD), the UK Government needs to implement policies designed to make major improvements to the energy efficiency of new and existing domestic buildings over the coming decades (HM Government, 2010; EU, 2011; DECC, 2014). With existing housing projected to account for approximately 70-80% of the housing stock in 2050, (Boardman, 2008; Palmer and Cooper, 2011), proposals suggest that these dwellings should undergo extensive retrofitting, with the installation of insulation, more efficient heating systems, and an increase in air tightness (DECC, 2014). However, in complex systems such as housing, policy formulation processes that are narrowly focused on single objectives (in this case climate change mitigation) while taking inadequate account for the complex and dynamic inter-relationships between objectives and outcomes, inevitably lead to a wide range of unintended consequences arising from both policy framing and implementation (Davies and Oreszczyn, 2012). To date, there has not been sufficient research that examines these unintended consequences – either positive or negative - that may impact building fabric, human health and wellbeing, the local and wider society and the environment.

One prominent consequence of energy efficiency modifications to the existing housing stock is the likely change in Indoor Air Quality (IAQ) (Wilkinson et al., 2009), which will in turn influence personal exposure to indoor airborne pollutants such as particulate matter (PM), a pollutant with known negative impacts on human health. Epidemiological evidence shows that the smaller fractions (aerodynamic diameter of 2.5 microns or less - $PM_{2.5}$) are particularly harmful to population health (WHO, 2005; COMEAP, 2009), with both short and long term exposure linked to a decrease in life expectancy and an increase in morbidity and mortality (WHO, 2006; COMEAP, 2009). The most serious health problems occur among susceptible groups with pre-existing lung or heart disease, along with the elderly and children (McMurry et al., 2004). There is a change in the relative risk of all-cause mortality of 6% per 10μg/m3, change in annual average $PM_{2.5}$ concentrations and a specific increase of 8% for cardio-pulmonary with 9% for lung cancer (Pope et al., 2002; 2004). There is no known 'safe' level of $PM_{2.5}$ and there will continue to be health risks associated with any exposure (WHO, 2006; DEFRA, 2013). However, no legislation or policy currently takes into account $PM_{2.5}$ exposure within the home despite research showing that $PM_{2.5}$ is a significant health issue in the UK (PHE, 2013). The 2011 fraction of mortality attributable to particulate air pollution is estimated to be 5.4% nationwide (based on outdoor $PM_{2.5}$ exposure), representing in excess of 24,000 deaths in 2011 (ONS, 2012).

Concentrations of $PM_{2.5}$ in domestic dwellings are affected by the infiltration of outdoor particles, emissions from indoor sources and the removal from the internal air by deposition and exfiltration, though some re-suspension (often related to domestic activities and general movement) also occurs (Gehin et al., 2008). Internally generated $PM_{2.5}$ has been linked to transient internal sources such as construction materials, fixtures and fittings, and appliances as well as intermittent emissions such as the burning of fuels and candles, smoking, cooking, heating and human domestic activities (Milner et al., 2005; Weschler, 2009). Individuals in the UK spend a large proportion of their time in indoor environments; a study of activity patterns in Oxford found participants were spending 95.6% of their time in indoor environments, with 66% of this time spent in their homes (Schweizer et al., 2007). Consequently, any changes in domestic indoor air quality (IAQ) following energy efficiency measures are likely to impact on population health (Wilkinson *et al*., 2009). Therefore, methods need to be developed to more fully understand and quantify the range of outcomes of this unintended consequence to more accurately predict the impact on indoor $PM_{2.5}$ concentrations in homes, personal $PM_{2.5}$ exposures and the impact on population health. The current research gap lies in assessing the existing concentrations in the housing stock. Furthermore, the impact of energy efficient refurbishment and ventilation interventions on both current and future indoor exposures to $PM_{2.5}$ in homes, taking into account the impact of a variety of influencing factors.

As part of the "Pollutants in the Urban Environment" (PUrE) Intrawise project that sought to develop a decision-support framework for a more sustainable management of urban pollution and using a modelling approach this thesis seeks to clarify how carbon emission reduction policies via the application of energy efficiency and ventilation measures (applied to dwellings in the UK) will influence the concentrations of $PM_{2.5}$ in the indoor environment (PUrE, 2012). In addition, it investigates the influences of building location, tenancy type, occupant income and behaviour, all factors that contribute to exposure levels. The results obtained will help inform both IAQ policies and refurbishment strategies by offering insights into the unintended consequence of the application of energy efficiency measures in highlighting the trade-offs between health and the decarbonisation of the domestic stock. For example, greater airtightness in buildings may reduce energy use and exposure to $PM_{2.5}$ from outdoor sources, but could increase concentrations of indoor $PM_{2.5}$ sources. This delicate balance can depend on a number of factors, which are discussed later. Selecting the optimal strategy can help minimise any negative impacts and maximise energy and health co-benefits. By developing a methodological framework, the modelling can also be used to investigate a wider range of airborne pollutants in different locations in the UK or elsewhere, where sufficient empirical input data exists.

1.1 Basis for Research

The exposure to indoor $PM_{2.5}$ from both indoor and outdoor sources and therefore the potential health impact experienced by residents depends on a complex range of interacting influences. These include for example; external pollutant concentration, meteorology, internal sources, building geometry, location and ventilation characteristics, tenancy type, occupant activity and income. In order to understand the various factors that influence personal $PM_{2.5}$ exposure by monitoring would require extensive use of a variety of expensive measuring equipment and complex methodologies to capture all the components and combine them. Alternatively, a modelling approach incorporating empirical data could be used and may be the only alternative when exploring future scenarios where additional inputs such as future climate change, the use of forced ventilation, alterations in building thermal characteristics and increased air tightness need to be taken into consideration (Johnson, 2001; Gerharz et al., 2009). By establishing how these multiple factors interact, it should be theoretically possible to determine the key influences on $PM_{2.5}$ concentrations in domestic properties and infer exposures, both currently and in a future low-carbon domestic stock. However, evaluation of these multiple influences is not a simple task- consequently, very little all-inclusive research has been carried out. In addition, some of the variables and mechanisms have previously proved difficult to quantify (Kauhaniemi et al., 2008; Gerharz et al., 2009).

Modelling work by Wilkinson, et al., (2009) using limited stock profiling for the UK, inferred that, in general, energy efficiency interventions may lead to a reduction in indoor domestic exposure to $PM_{2.5}$ and are therefore positive for health. However, this result was strongly influenced by, for example, the degree of airtightness, the choice of energy intervention and ventilation system, and the strength of $PM_{2.5}$ sources and sinks. Other factors shown to have impacts, such as building location and external pollutant sources (Vardoulakis et al., 2007), along with the importance of occupant behaviour (Baxter et al., 2007) were not investigated, nor the potential variation of impacts across different tenancy and income groups (Fabian et al., 2012). For the purposes of this research, the most relevant aspects of occupant behaviour are considered to be, times of cooking, movement/location of individuals within the properties, window opening and the execution of domestic activities e.g. cleaning, sweeping etc. These are in turn affected by complex interaction with building characteristics and individual and/or household characteristics, such as income and tenure. Whilst Wilkinson, et al. (2009) highlighted a potential issue with intervention policy affecting indoor $PM_{2.5}$, its narrow range of empirical inputs made its conclusions uncertain. For example, a single indoor source for $PM_{2.5}$ (from cooking) was used, whereas multiple emission sources are known to occur including re-suspension of previously deposited material Ozkaynak et al. (1996); He et al. (2004) and Afshari et al. (2005). In addition, the study was conducted prior to publication of changes in Approved Document L for increased airtightness as a result of energy efficiency targets and Approved Document F (targeting adequate ventilation), both in October 2010, affecting all future building work. Finally, no sensitivity analysis was undertaken and therefore the range and distribution of errors in the final results are unknown. Other studies such as Fabian et al. (2012) have considered multiple sources of $PM_{2.5}$, on limited existing stock profiles but not the consequences of energy efficiency applications or ventilation strategies, although they have suggested a possible link between income and tenancy type on indoor $PM_{2.5}$ exposure. They also conclude that over simplified one-compartment box models mischaracterise concentrations and source contributions, implying the need for more complex modelling to improve accuracy.

Figure 1.1 highlights the main variables involved when trying to establish personal indoor $PM_{2.5}$ exposure both currently and in a future low-carbon domestic stock. It illustrates that a complex relationship exists between the influencing factors, with a gap in current research that requires far more detailed investigation. This will yield a more robust and accurate profile of personal indoor domestic

$PM_{2.5}$ exposures in England, to assist in quantifying the potential health impacts of this unintended consequence resulting from climate mitigation interventions on the domestic building stock. In addition, the same methodology could be used with other airborne pollutants to evaluate their impact.

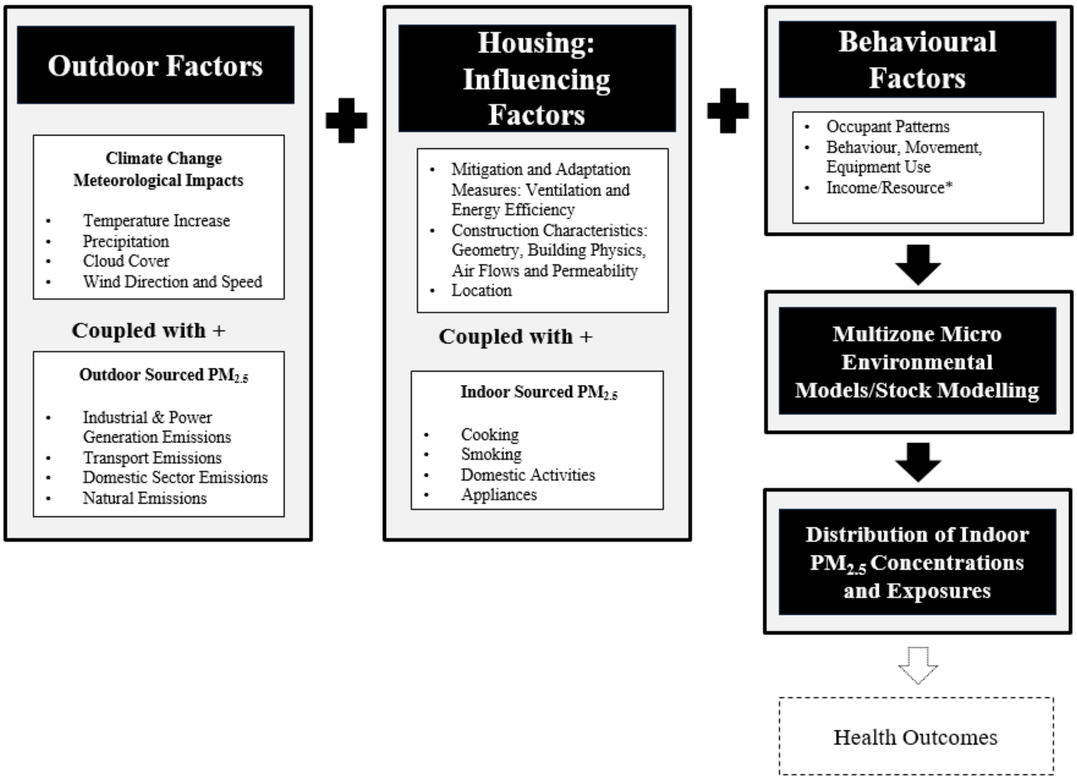

Figure 1.1 The range of direct and indirect factors influencing indoor domestic $PM_{2.5}$ concentrations and exposure and their subsequent input to modelling software.
*Income/Resource as an impact on indoor concentrations of $PM_{2.5}$ is explicitly shown in Figure 7.1 and covered in chapter 7

In Figure 1.1, outdoor data; for example, empirical external $PM_{2.5}$ concentration and meteorological inputs may be obtained from the UK Automatic Urban Rural Network (AURN) (DEFRA, 2010), or the DEFRA mapping project can be used where insufficient monitoring stations exist (DEFRA, 2013). However, these may not be representative of personal exposure experienced by individuals due to the high spatial variability of urban air pollution and the differences between indoor and outdoor concentrations (Levy, et al., 1998). As a result, a more detailed evaluation is necessary to quantify all variables at each step of the pathway in order to include the links between outdoor factors and $PM_{2.5}$ sources, the mitigating impact of housing and indoor $PM_{2.5}$ sources and how these may influence personal indoor exposure. This raises a number of specific questions that need to be addressed regarding $PM_{2.5}$ concentrations in domestic properties, such as:

- What is the impact of building characteristics and envelope properties in influencing indoor domestic $PM_{2.5}$ exposures from indoor and outdoor sources?
- Will the application of retrofit energy efficiency and ventilation strategies reduce indoor $PM_{2.5}$ exposures in homes?

- How will making buildings more airtight in order to prevent ventilation heat loss, impact on indoor $PM_{2.5}$ exposures from indoor and outdoor sources?
- How does occupant activity/behaviour in homes influence personal $PM_{2.5}$ exposure?
- Does the geographical location of similar properties lead to a variation in indoor domestic $PM_{2.5}$ exposure?

Understanding the answers to these questions will assist with:

- A clearer understanding of building characteristics that influence indoor $PM_{2.5}$ concentrations.
- A better understanding of retrofitting interventions that yield co-benefits of a reduction in $PM_{2.5}$ exposures and those that lead to greater levels of exposure.
- Prioritising those properties and locations subject to highest exposure risks and having the potential for greatest GHG reductions.
- Understanding which aspects of occupant behaviour may affect $PM_{2.5}$ exposure, with a view to then target these behaviours via tailored behaviour change interventions.
- A more informed conclusion regarding the possible trade-offs between climate change mitigation goals for housing and human wellbeing around $PM_{2.5}$ exposures.

This thesis firstly examines the nature and causes of unintended consequences in relation to the housing stock. Having established changes in indoor air quality (IAQ) and specifically $PM_{2.5}$ as a consequence of note, it considers how multizone airflow and contaminant transport analysis software can be used to model the changes in $PM_{2.5}$ concentrations in homes. By combining both existing and adapted modelling techniques, this thesis aims to address the unintended consequences of the application of energy efficiency and ventilation interventions on the housing stock alongside the range of influences on personal indoor $PM_{2.5}$ domestic exposure. This leads to the questions underpinning this study:

1.2 Research Questions

The concentrations of $PM_{2.5}$ within homes will be dependent on a variety of sources; infiltration of outdoor particles, emissions from indoor sources and the removal from the internal air by deposition and exfiltration. These in turn will be influenced by a range of mitigating factors such as building permeability, ventilation rates, external meteorology as well as occupant behaviour, income and location. The application of energy efficiency and ventilation interventions to achieve climate change mitigation goals adds a further layer of complexity.

This thesis examines these complex and dynamic influences that contribute to modelled indoor $PM_{2.5}$ exposure and addresses the key or core research question:

- *Will the introduction of climate change mitigation strategies on dwellings lead to negative unintended consequences by increasing $PM_{2.5}$ concentrations in English dwellings, or provide co-benefits for health by a reduction of $PM_{2.5}$;*

In order to clarify the key question, this research will also consider the following subsidiary questions;

- *What are the factors that influences these concentrations and can their contribution be quantified by the use of modelling software?*
- *Whether occupant behaviour, income, tenancy or location have a modifying influence on personal indoor domestic $PM_{2.5}$ exposures, and if such impacts are greater than the uncertainties in the models used to calculate such changes?*
- *Whether such models be constructed with different housing stock/ventilation profiles and other airborne pollutants such that they may have a wider use within other research projects?*

1.3 Research Aims and Objectives

Following a review of the nature and scope of unintended consequences in relation to energy efficiency and ventilation strategies on the English housing stock; existing knowledge surrounding the modelling of $PM_{2.5}$ and the many factors contributing to indoor exposure and in order to investigate the research questions, a number of aims have been set and with these a series of objectives proposed to achieve those aims as seen in Table 1.1. Figure 1.2 represents a flow diagram of how the investigations proceeded in relation to the aims and objectives of the thesis.

Table 1.1 The aims and associated objectives of the thesis

	Aim	Objective
1	Understand the context and causes of changes in indoor $PM_{2.5}$ concentrations	Via a literature review, scope the causes and domains of impact of unintended consequences resulting from policies promoting the application of a variety of energy efficiency measures to housing.
2	Develop a robust methodology for modelling indoor domestic $PM_{2.5}$ exposures	Investigate the range of methods used for modelling indoor $PM_{2.5}$ and the factors influencing exposure. Evaluate previous works and its applicability to the thesis questions.
3	Carry out a preliminary analysis of the role of building characteristics and climate change mitigation strategies on indoor domestic $PM_{2.5}$ exposures and the impact of occupant behaviour.	Using CONTAM and modelling the current Greater London Authority (GLA) housing stock, investigate the application of a specific energy efficiency intervention (air-tightness with MVHR). Calculate post intervention $PM_{2.5}$ concentrations and quantify impacts for the GLA. Post process results for different occupant behaviours and activities.
4	Estimate the uncertainty in key variables impacting estimates of indoor domestic $PM_{2.5}$ exposure and their distributions.	Investigate the key influencing variables on indoor $PM_{2.5}$ concentrations and uncertainty within the CONTAM models and applied at stock-level using differential sensitivity for both input variables and computational processes.
5	Assess the impact of geographical location and spatial factors as influences on indoor domestic $PM_{2.5}$ exposure.	Using The SCRIBE tool (incorporating CONTAM) and modelling the current London and Milton Keynes housing stocks; apply a variety of energy efficiency and ventilation interventions and, investigate the potential for achieving climate change targets and the impacts on indoor $PM_{2.5}$ concentrations in two different locations.
6	Investigate the influence of energy-efficient retrofits on indoor $PM_{2.5}$ concentrations for different income groups and tenancies across England.	Using EnergyPlus and its GCM model building archetypes representative of the current and post retrofit English Housing stock to predict indoor $PM_{2.5}$ exposures from both indoor and outdoor sources. Using statistical analysis (ANOVA) to investigate differences between the various tenancies and income groups.

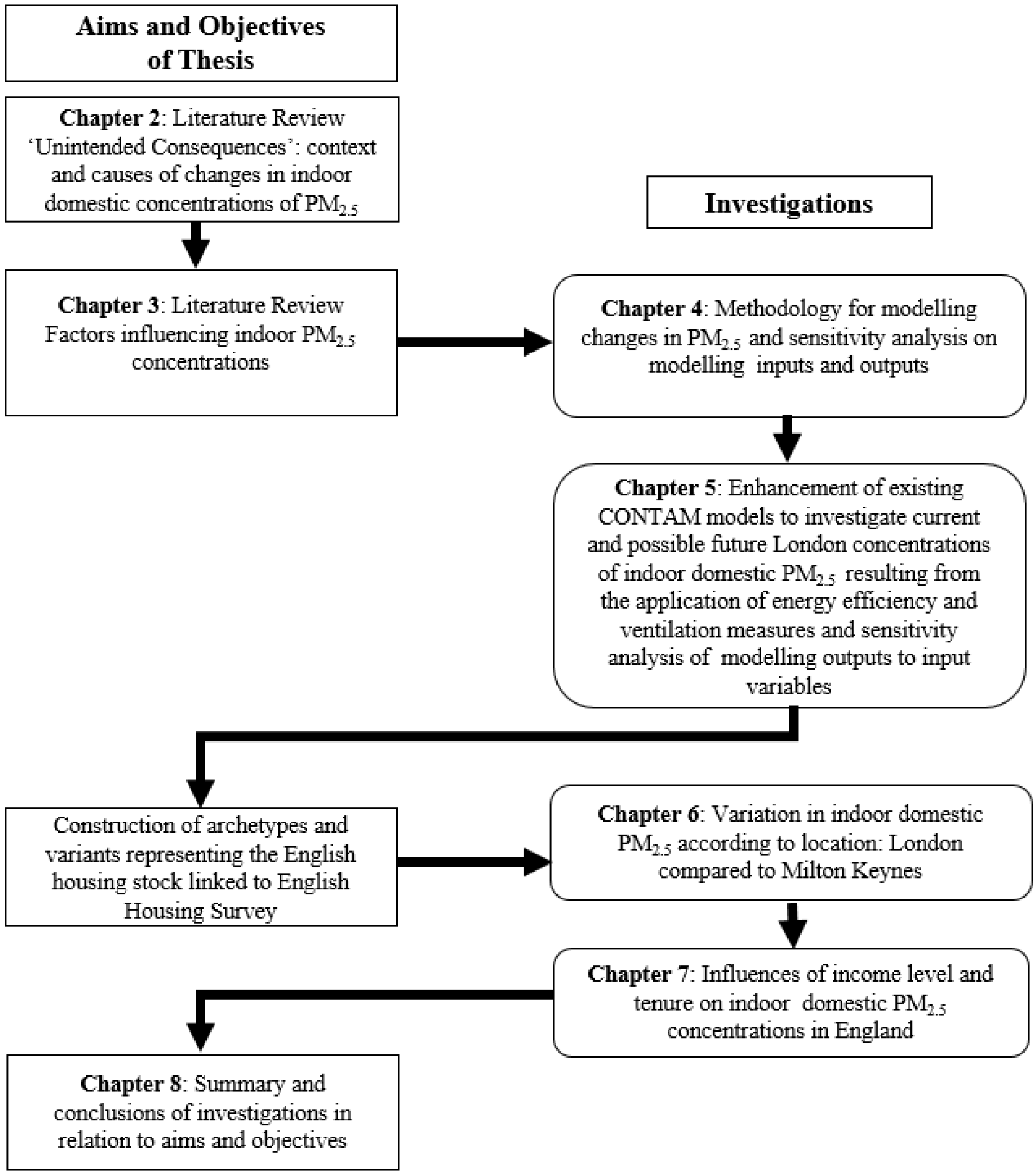

Figure 1.2 Flow diagram of investigation

These research aims and objectives have been successfully achieved and the methodology used has been shown to be applicable to obtaining both $PM_{2.5}$ and other indoor domestic airborne pollutant concentrations in different building archetypes with various energy efficiency and ventilation interventions, different locations, occupant groups (tenure and income) and behaviours.

1.4 Research Tools

Having established changes in indoor domestic concentrations of $PM_{2.5}$ as a notable unintended consequence of policies to improve the energy efficiency of the English Housing stock, an investigation of possible tools was conducted. There are a variety of computer-based exposure models available to deal with the complex scenarios seen including statistical regression, micro-environmental (mE) and computational fluid dynamic (CFD) models (Milner et al., 2010). The complexities in this study lend themselves to the use of micro-environmental (mE) modelling software which enables the simulation of a variety of domestic geometries and the incorporation of inputs from multiple disparate sources. Model selection includes the identification and quantification of any influences that will impact indoor domestic exposure to $PM_{2.5}$. Following investigation based around the scale of this study its multiple objectives range of disparate inputs required; a number of models were chosen based on their applicability, requirements and limitations as described in section 4.4. These include:

(1) A multizone micro-environmental (mE) model - CONTAM which has been extensively validated - was chosen for this study (Haghighat, 1996; Emmerich, 2001). This programme developed by Building and Fire Research Laboratory of the National Institute of Standards and Technology (NIST), can be used to study Indoor Air Quality (IAQ) simulating the effects on pollutant concentration of ventilation, permeability and air movement in multizone models (Dutton, et al., 2008).

(2) EnergyPlus, a validated energy analysis and thermal load simulation program developed by the U.S Department of Energy (EP, 2015). The recent addition of a module to this programme: the Genetic Contaminant Module (GCM) has enabled the study of an individual pollutant, its movement and subsequent concentrations within a building modelled in EnergyPlus (Taylor et al., 2014a).

(3) In addition, to achieve the study objective of comparison between different locations, a UK housing stock computer modelling programme 'Strategies for Carbon Reduction in the Built Environment' (SCRIBE) was further developed and used, based on work by Hamilton et al. (2012, 2015) and described in section 6.1.3. This incorporates (i) a building stock component, (ii) an energy efficiency module, (iii) a validated airflow and pollutant transport component and (iv) a module that calculates changes in annual energy use under a range of both housing interventions and changing electricity grid carbon intensities and consequent CO_2 emissions.

It is acknowledged that all models, however detailed, are simplifications of reality and there are differences between actual pollutant movement and concentration in a real domestic property when compared to computer models (Emmerich, 2001). Sensitivity analysis on inputs to the models is therefore needed to examine the sensitivity of the results to individual model inputs and assumptions, whilst comparisons with empirical studies are needed to confirm accuracy (Lomas and Eppel, 1992). In addition, statistical methods have been used to post-process some results and investigate differences for example in tenancy types and income groups.

Whilst the methods developed in this thesis are applied to particular locations they are transferrable to other contexts with different airborne pollutants.

1.5 Originality and Novelty

This thesis scoped the range and domains of the impact of unintended consequences resulting from of the application of energy efficiency and ventilations measures on the English housing stock, developing a series of simulations to investigate in-depth the changes in indoor domestic exposure to $PM_{2.5}$. These were subsequently used to ascertain the impacts on changes in concentrations for other airborne pollutant concentrations. It required a multi-disciplinary approach using, micro-environmental, pollutant modelling and energy analysis software. This represents the first time a thorough investigation of the multiple influences on indoor domestic $PM_{2.5}$ concentrations have been carried out in the context of climate change and mitigation measures, which fills an important research gap. Although the work was carried out as a team on a number of projects where the author acted as a researcher, the major contributions for which he was personally responsible (i.e. are exclusively his) cover (1) contribution to methods/tools and (2) direct contributions to knowledge.

(1) Contributions to Methods/Tools:

- The creation of an enhanced, transferable methodology for modelling domestic stock profiles in CONTAM, enabling the production of multiple geometries for building and systems able to comply with changing Building Regulations. These models enable investigation of a variety of future mitigation measures and airborne pollutant types.

- The production of complex models in CONTAM representing the English housing stock, linked to the English Housing survey, and able to predict occupant exposure to various airborne pollutant concentrations using a variety of energy efficiency and ventilation interventions and currently used within the Department of Energy and Climate Change 'Health Impacts of Domestic Energy Efficiency' (HiDEEM) model (Hamilton et al., 2012, 2015), to monetise the health impacts of energy efficiency interventions.

(2) Contributions to Knowledge:

- The first attempt to characterise the unintended consequences of policies to reduce end-use housing energy demand and to highlight among other outcomes the changes to indoor air quality and $PM_{2.5}$ exposure. The research paper resulting from this work was awarded 'Paper of the year 2014' by Indoor and Built Environment, a peer reviewed journal.

- Research filling a knowledge gap enabling quantification of possible changes in future population $PM_{2.5}$ exposure, and a range of other airborne pollutants and subsequent health impacts, by modelling current and possible future stock profiles under a variety of mitigation scenarios for England.

- The creation of $PM_{2.5}$ exposure profiles able to be used in Health Impact Assessment (HIA) software to quantify the health outcomes of the various strategies, enabling policy makers to see the health consequences and costs of various energy efficiency and ventilation strategies.

- The first execution of differential sensitivity analysis on housing stock modelling in order to quantify and understand the uncertainties within the modelling and the key variables impacting indoor domestic $PM_{2.5}$ concentrations and their distribution.

- Quantification by modelling, of the relative impacts of different locations and its influence on indoor $PM_{2.5}$ concentrations in the housing stock including the enhancement of the 'Strategies for Carbon Reduction In the Built Environment' (SCRIBE) tool, to incorporate data for London and Milton Keynes

- A simple post-processing method for determining the relative exposures to $PM_{2.5}$, experienced by individual occupants' dependant on their location and activity within a property, highlighting the impact of differences in behaviour.

- Producing research linking income, tenancy and the impacts of energy efficiency interventions on indoor $PM_{2.5}$ exposure in homes thereby filling an important knowledge gap.

1.6 Specific Contributions by Others

Some of the research presented within this thesis was not carried out in isolation, but as part of a team of researchers on several different projects. Various people contributed to these projects and publications, but this thesis presents work where the author was the main lead and/or provided significant or exclusive contribution. A list of specific contributions by others is provided below. The original contribution to knowledge arising from this thesis was articulated in the previous section. It should also be acknowledged that some aspects of the work presented in this thesis build upon the work of others, especially the stock modelling approach (Wilkinson et al., 2009; Oikonomou et al., 2012; Taylor et al., 2014a, 2014b).

- ***Chapter 2 - An Investigation of Unintended Consequences of Energy Efficient Refurbishment of the Housing Stock:*** Some guidance and suggestions for investigation and framing were received from Neil May MBE., Prof. Mike Davies (UCL) and Dr Alex Macmillan (now University of Otago, NZ).
- ***Chapter 5 - Predicted Changes in Indoor $PM_{2.5}$ in London's Domestic Stock, due to Energy Efficiency Retrofits: An Idealised Case:*** The original stock models used (prior to updating by the author) were supplied by Dr Ian Ridley (now University of Hong Kong), who also assisted with checking outputs. Outputs from OSPM were supplied by Dr Sotiris Vardoulakis (now

Institute of Occupational Medicine) using data supplied by the author. In order to speed up computational times an Excel macro 'CONTAM-batch' (see Appendix F for details) was developed in conjunction with Dr Phillip Biddulph and run using Strawberry Perl, 2008.

- ***Chapter 6 - Variation in Indoor $PM_{2.5}$ Exposure Between Locations in England: London and Milton Keynes, A More Realistic Case:*** This investigation used a version of the HiDEEM tool known as SCRIBE. This tool contains a variety of inputs and modules. Those for energy use and health impacts were created by Dr Ian Hamilton (UCL) and Dr James Milner (LSHTM). The pollution components were constructed in CONTAM by the author using archetypes supplied by Dr Eleni Oikonomou (UCL) with assistance with input checking by Dr Ben Jones (University of Nottingham) and Dr Payel Das (University of Oxford). Additionally, Dr Das created the algorithms in Matlab within the SCRIBE tool to enable the calculation of Milton Keynes data and current and future carbon intensities (CI) using data table constructed by the author. Health data was supplied by Dr James Milner (LSHTM) using exposure outputs supplied by the author.
- ***Chapter 7 - Variations in Indoor $PM_{2.5}$: Exposure for Different Income Groups:*** this work examines the impacts on different tenures and income groups of energy and ventilation retrofitting measures on the English domestic stock and the consequent impacts on concentrations of indoor $PM_{2.5}$ exposure. Dr Payel Das (University of Oxford) and the author created the algorithms in MATLAB that enabled probabilistic analysis of impacts on $PM_{2.5}$ exposure for different income groups using data supplied by the author.

1.7 Thesis Structure

This section briefly outlines the contents of each chapter within the thesis:

- ***Chapter 1 - Background:*** Introduces the subject of the thesis, giving background details and describes why research in the area is necessary. It also states the specific research questions, the objectives of the study, and its originality.
- ***Chapter 2 - An Investigation of Unintended Consequences of Energy Efficient Refurbishment of the Housing Stock*** A scoping review setting the context for the study.
- ***Chapter 3*** **- Factors Influencing** ***Indoor Domestic $PM_{2.5}$ Exposure and Health:*** Contains a review of published research literature and its relevancy to the research project.
- ***Chapter 4 - Overview of Methodology:*** Outlines the general methodologies used in this thesis, including the choice of modelling tools used.
- ***Chapter 5 - Predicted Changes in Indoor $PM_{2.5}$ in London's Domestic Stock, due to Energy Efficiency Retrofits: An Idealised Case:*** Contains a preliminary analysis of the many

influences determining indoor $PM_{2.5}$ concentrations in London dwellings. It also investigated modelling uncertainties through sensitivity analysis on key variables.

- ***Chapter 6 - Variation in Indoor PM2.5 Exposure Between Locations in England: London and Milton Keynes, A More Realistic Case:*** Examines the impact of location and spatial factors on $PM_{2.5}$ concentrations, by contrasting the housing stocks of London and Milton Keynes.
- ***Chapter 7 - Variations in Indoor $PM_{2.5}$: Exposure for Different Income Groups:*** Examines the impacts on different tenures and income groups of energy and ventilation retrofitting measures on the English domestic stock and the consequent impacts on concentrations of indoor $PM_{2.5}$ exposure.
- ***Chapter* 8** – ***Summary Discussion, Conclusions and Future Work:*** Articulates a summary of the main evidence arising from the thesis in relation to the research objectives and the limitations of the study and considers key findings of the thesis in relation to the stated objectives and draws final conclusions and suggestions for future work.
- ***Appendices*****:** Includes supplementary material with selected research papers where the author of this thesis is included.

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Chapter 2

An Investigation of Unintended Consequences of Energy Efficient Refurbishment of the Housing Stock

Introduction

The previous chapter established the need for a thorough investigation of indoor $PM_{2.5}$ exposure in homes. It outlined the study topic, the areas of examination and the order in which the research was to be conducted. The study has been shown to be multifaceted requiring a multidisciplinary approach in order to capture the complex range of factors influencing personal $PM_{2.5}$ exposure. The investigation in this chapter sets the context for the study by first considering the causes of unintended consequences emanating from policies to make the housing stock more energy efficient and highlighting changes in indoor $PM_{2.5}$ exposures as a notable issue requiring further examination. From this perspective, a description of $PM_{2.5}$ and the many factors influencing exposure to $PM_{2.5}$ are examined in detail in the following chapter.

In addition to two conference presentations, a research paper resulting from the investigations in this chapter was published in a peer reviewed journal and was awarded Indoor and Built Environment, Best Paper 2014 by Sage Publishing. Further research papers drawing on this study are listed in the 'thesis associated publications section' commencing on page 15.

2.1 Unintended Consequences: Definitions and Causes

Merton (1936), when discussing policy impacts, defined 'unintended consequences' as outcomes that arise unintentionally as a result of policy framing, development or implementation. Multiple direct and indirect consequences can occur. They can be broadly grouped into two categories: (i) an unexpected benefit or negative effect (or a combination of both), which may occur in addition to the desired effect of the policy or action; (ii) an effect contrary to the original intention that undermines the intention and even makes the problem worse. The complex interdependence of many of the consequences is discussed in detail later.

European and domestic legislation motivated by (GHG) reduction concerns aims to substantially improve energy efficiency in both new and existing UK homes in the coming decades (DECC, 2014). Existing dwellings are likely to represent 70 - 80% of the 2050 stock (Boardman, 2008; Palmer and Cooper, 2011) and are likely to undergo extensive retrofitting with a range of measures that will increase air tightness, insulation, glazing improvements and the efficiency of heating systems in order to help meet the UK's ambitious GHG reduction targets (80% of 1990 emissions by 2050) (Wilkinson et al., 2009). The summary of recent legislation and national policy in Table 1 demonstrated the Government's approach to GHG reduction involving the housing sector; with policies seeking to improve energy efficiency, reduce the carbon intensity of energy generation and change the energy related behaviour of building occupants (Mavrogianna et al., 2012; DECC, 2014). Currently, much of this legislation/policy has now been withdrawn with new legislation and policies pending. In addition, there is also huge uncertainty in light of the recent 'Brexit' vote and the UK's withdrawal from the EU as to whether current or future EU air quality standards will be adhered to and what the nature of future policy on housing will be. However, the general applicability/validity of the impacts seen are still relevant to the discussion as it stands and specifically changes in indoor concentrations of $PM_{2.5}$.

The need to consider the linkages that exist between buildings, human wellbeing, local and wider societal, and environmental impacts when forming such policies has been noted previously (Davies and Oreszczyn, 2012). Focusing on housing, this section of the review illustrates the complex nature and range of possible unintended consequences arising from policy framing and implementation that is limited to a focus on climate change mitigation. It seeks to exemplify and categorise the broad range of possible unintended consequences that may arise as a result of proposed energy efficiency measures. It further suggests the need for a broader approach to policy decisions that integrates multiple objectives about housing and includes consideration of a wider range of outcomes and involves multiple stakeholders in decision-making so that co-benefits may be optimised, negative impacts reduced and trade-offs made more explicit. One notable unintended consequence that will be explored in detail is the change in indoor air quality, specifically the concentrations of $PM_{2.5}$ following energy efficient refurbishment. The review will establish the complex and dynamic range of factors that combine to influences personal indoor $PM_{2.5}$ exposure and how they can be modelled.

Table 2.1 Summary of UK legislation, policies and incentives that have been used to promote the decarbonisation of the housing stock in place at the time of the study.

Legislation	Description
Climate Change Act 2008	Requires emissions reductions of 80% by 2050, introduces legally binding carbon budgets and sets a legal framework for climate change adaptation.
Energy Bill 2012	Electricity Market Reform including predictable incentives for investment in low-carbon generation (Contracts for Difference) and ensuring adequate supply (Capacity Market).
Building Regulations and associated technical guidance (ongoing)	Includes legislative requirements for energy efficiency and GHG emissions from new buildings as well as requirements for retrofitting existing buildings for any reason.
Policies and Incentives*	Description
Renewables Obligation (2002-date)	Requirement for electricity suppliers to source an increasing proportion of electricity from eligible renewable sources or pay a penalty. Suppliers buy certificates from generators and present them to the regulator or buy-out their obligation.
The Green Deal (2012-2015)	This was the main national incentive for retrofitting existing dwellings, including a loan scheme covering loft and external wall insulation (including solid and cavity walls); boiler upgrade or replacement with heat pump; renewable energy generation (solar panels or wind turbines); double glazing and draught proofing. Expected financial savings must be equal to, or greater than, the costs. Loans were attached to property utility bills.
Energy Company Obligation (ECO) (2013-2017) To be replaced by alternative scheme	Requirement for Energy Companies to fund energy efficiency improvements under three obligations: (i) provision of insulation to low income households in specific target areas; (ii) provision of heating and insulation for beneficiaries in private tenure and (iii) installation of less cost effective measures not meeting the financial savings requirement of the Green Deal (e.g. solid wall insulation). Energy companies are expected to respond to these obligations by increasing energy prices.
Feed-in Tariff (FITS) (2010-date)	Guaranteed payment from electricity suppliers for surplus electricity from small-scale (less than 5MW), low-carbon generation – under review.
Domestic Renewable Heat Incentive (RHI)	Proposed future extension of the non-domestic RHI to houses, providing financial support for installation of eligible technologies (e.g. biomass boilers, ground source heat pumps, solar thermal).

*As previously noted, whilst some policies/incentives have been abandoned and new policies have yet to be formulated, their influence or that of similar future policies are still relevant to this study. Additionally, the restructuring of DECC (within BEIS) and the forthcoming impacts of 'Brexit' are as yet unclear.

2.2.1 Review Framework

In the absence of a specific published structure for the investigation of potential relationships between energy efficiency, housing, people and nature, a broad exploratory framework was used (Figure 2.1) to define domains of possible consequences (Barton and Grant, 2006). This framework was originally designed to illustrate the relationships between health and wellbeing in neighbourhoods and the physical, social and economic environment, but is considered a valuable holistic model that provides a useful lens to direct the areas for literature search by revealing the multiple domains of consequences of policies to improve energy efficiency.

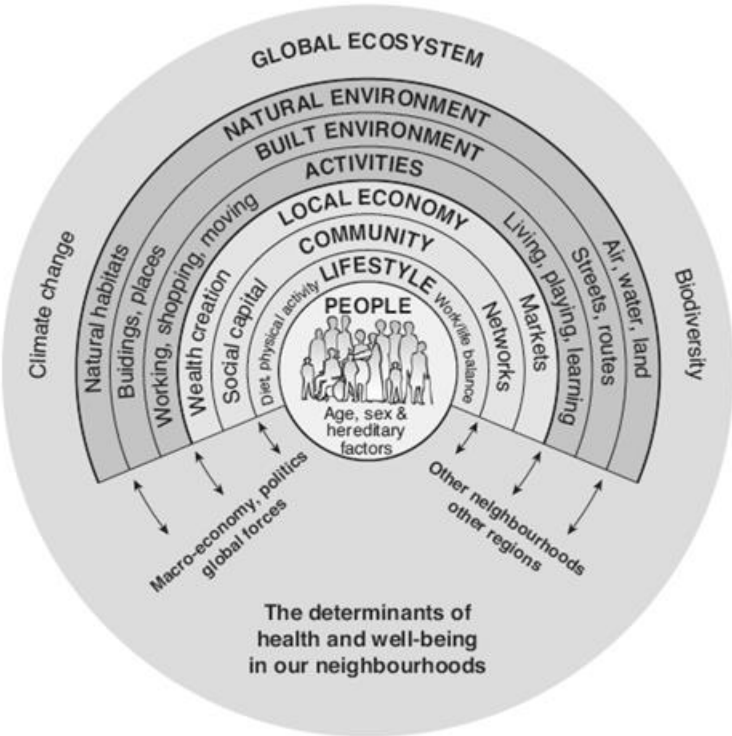

Figure 2.1 Holistic framework of health and wellbeing (Barton and Grant, 2006) adapted from (Whitehead and Dahlgren, 1991).

Using the framework described above, a scoping search of the literature was conducted across the following disciplines: building physics; construction technology and practices; health and wellbeing; and social sciences. The search method used is shown in section 2.1.2. Using the framework domains, an initial set of keywords were developed for each energy efficiency intervention and further used in combination with outcomes relevant to that intervention, for example human health. An example is shown below in Table 2.2. The full range of search terms are shown in Appendix A. Additional terms and combinations revealed by the literature search were also investigated.

Table 2.2 Example of keywords used in the literature search

Policy Impact	Initial Keywords	Domain combination	Additional Revealed terms
airtightness	permeability, airflow, air change rate, indoor air, indoor air quality, airtight	health, well-being, consequence	mental health, physical health, psychological well-being, child development

The impacts of the range of interventions on dwellings were considered independently so as to reveal the pathways of their individual consequences. Themes emerged from the literature which led to specific interventions being investigated including: increasing airtightness, purpose provided ventilation (PPV); insulation (including impacts of double glazing) and impacts related specifically to 'traditional built' structures as opposed to new builds, due to their constructional differences (STBA, 2013). Additional areas of investigation include the implications of the policy funding structure under the Green Deal; the UK Coalition Government's flagship carbon emission reduction policy for domestic properties (DECC, 2014), as well as the potential effects of changes to design, construction and manufacturing processes that may result from current policy.

2.1.2 Selection Criteria and Analysis

The search was limited to studies in English published from 1990-2014. Studies are included that make a direct connection between an intervention to reduce GHG emissions from, or improve the energy efficiency of, dwellings and an impact on one or more domains described in the framework above. Studies that failed to meet these criteria are considered not relevant to the scoping review and were rejected. The findings of the included studies are used to group and characterise described relationships between interventions and outcomes. These relationships are tabulated, summarising the short pathways described in the studies between the impacts on buildings, people and the natural environment. Where there is unresolved debate about the direction of effects of an intervention on an outcome, both theories are included. Although greater emphasis is placed on systematic reviews of particular effects of interventions on housing, the aim was not to assess the quality of the evidence, nor to report on relative effect sizes or the strength of relationships.

2.2 Results

Nearly 1600 potentially relevant studies were identified. Of these, 436 had content relevant to this study, and of these 206 met the inclusion criteria as defined above. 119 unintended consequences were highlighted, representing the impacts related to the application of the investigated energy efficiency policy measures. However, many individual consequences further impact on multiple domains resulting in a total of 196 possible outcomes reported across the studies. The papers reported impacts across many of the domains identified by the reference framework used (Figure 2.1) including the built environment, life style, and activities, community, local economy, the natural environment and the wider global ecosystem. It also identified some intervention effects that did not fit well within the holistic framework, including new legal ramifications and impacts on household-level economics. These have been included in the results and indicate potential future additions to the framework. The included studies described the effects of interventions that could be categorised as impacts associated with:

- increasing dwelling airtightness;
- replacing uncontrolled ventilation with purpose provided ventilation;
- insulating properties and raising indoor temperatures (this issue is not discussed here).

Although pertinent to unintended consequences, it is not a key issue in respect of $PM_{2.5}$, except in relation to airtightness, (which is dealt with).

A further set of unintended consequences have been reported that relate to current options for funding interventions and to the way that such interventions are being implemented. Within these categories, many studies also explored the particular impacts on older/traditional houses compared with more modern ones due to their constructional differences. The term 'traditional' is generally used to define a structure built prior to 1919 with solid walls constructed with moisture-permeable materials (Historic Scotland, 2007; STBA, 2012). Such buildings are estimated to represent almost a quarter of the current UK housing stock. They have specific issues different from the rest of the built stock for example; heat loss and moisture movement in solid walls (Historic Scotland, 2007; STBA, 2012). Both current regulations and the Green Deal and related policies did not take these differences into account when applying efficient technologies, although work is currently underway to address some of these issues (STBA, 2012). Due to the substantial range of consequences uncovered, it has not been possible to capture all individual impacts in any depth within this review; however, the following sections demonstrate the level of detail that exists for some known consequences.

2.2.1 Impacts Associated with Increasing Dwelling Airtightness

Studies described the airtightness impacts of a range of measures including for example; draft-proofing, the provision of double glazing, insulation of loft spaces and the filling of cavity walls/solid wall insulation/external wall insulation. For these interventions a range of both positive and negative impacts on a range of domains were described. Increases in airtightness of dwellings should result in reduced ventilation heat loss through lowered air change rates potentially leading to reduced energy consumption and GHG emissions (Das et al., 2013). The potential for reduction of noise ingress from outside created by these measures can have further impacts, such as a more peaceful atmosphere and the accompanying sense of security, which has a positive impact on mental health and psychological wellbeing (Sanz et al., 1992; Van Kempen et al., 2012). Improvements in child development in the spheres of physical, social and emotion health as well as behavioural outcomes are reported (Laventhal and Newman (2010). These positive impacts have been attributed to the 'reduction' in noise (Evans, 2003); conversely it has been emphasised that the 'absence' of sound (e.g. sounds from nature) may lead to negative mental health impacts (Evans, 2003; Van Kempen et al., 2012). For some individuals, this can lead to anxiety from both real and perceived threats (Lorenc et al., 2012) and a possible sense of isolation and disconnection having further impacts on social cohesion. Increased window opening to compensate for lack of natural sounds could lead to increases in ventilation heat loss working against GHG emission reduction (Fabi et al., 2012).

External sealing of the building envelope to increase airtightness was found to have the additional benefit of making properties more watertight and is recommended as a climate change adaptation measure thereby reducing possible future impacts from excess rainfall and the likelihood of water damage and mould/rot risk (Williams et al., 2012). However, other authors have described links between lower air change rates and a rise in relative humidity (RH), leading to increases in house dust mites, mould, severity of asthma and allergies (Viitanen et al., 2009; Ucci et al., 2011). Also in fabric decay in existing properties, particularly traditional buildings (STBA, 2012). Further rises in RH are produced when clothes are dried indoors and have been linked to increased exposure to microbiological pathogens and infectious diseases (Porteous et al., 2012). In new builds, with tighter construction drying out times for ‘wet trades’ are extended leading to higher RH over a prolonged period during initial occupancy (HM Government, 2010).

Other changes in indoor air quality were also identified as a further consequence of the lower air change rates, beyond those associated with increased humidity. Whilst a reduction in pollutants from external sources such as $PM_{2.5}$ which has known negative health impacts is noted (Wilkinson et al., 2009), an increase in exposure to indoor sources of pollutants such as $PM_{2.5}$, volatile organic compounds (VOCs) and environmental tobacco smoke (ETS) may occur (Wilkinson et al., 2009; Crump et al., 2011; Yu and Kim, 2011). There is also emerging modelling evidence for a population-wide increase in cancer risk from increased exposure to radon indoors, an airborne pollutant known to be carcinogenic (Das et al., 2013; Milner et al., 2014).

Examples of these relationships between increasing airtightness and human and environmental wellbeing are summarised in Table 2.3, which demonstrates the method used to map the pathways described between interventions and individual unintended consequences. The impacts on indoor $PM_{2.5,}$ the subject of this study are shown in italics (point 9).

Table 2.3 Examples of unintended consequences arising from the application of energy efficiency measures; airtightness

A	B	C	D	E	F	G	H
No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
1	Airtightness		Quieter Environment	Peace/Wellbeing / Security	Mental Health Psychological Well Being	+	Sanz et al., 1993. Van Kempen 2012 **D,E,F**
2	Airtightness		Quieter Environment	Isolation/ Disconnection	Mental Health Psychological Well Being	–	Lorenc et al., 2012 **D,E,F**
3	Airtightness		Quieter Environment	Anxiety: real and perceived threats	Psychological Well Being	–	Lorenc et al., 2012 **D,E,F**
4	Airtightness		Quieter Environment	Reduction in Noise	Mental Health	+	Evans, 2003; Kempen et al., 2012. **D,E,F**
5	Airtightness		Quieter Environment	Absence of sound	Mental Health	–	Evans, 2003; Kempen et al., 2012. **D,E,F**
6	Airtightness		Quieter Environment	Improvements in physical health; social, emotional, and behavioural outcomes	Child Development	+	Leventhal and Newman 2010 **D,E,F**
7	Airtightness	Lower air change rate	Increased RH Timber decay	Increase in HDM and mould, severity of asthma and allergies.	Physical Health	–	Viitanen et al., 2010; Ucci et al., 2011 **B,C,D,E,F**
8	Airtightness	Lower air change rate	Increased RH Clothes drying issues	Increase in and exposure to microbiological pathogens. And infectious diseases	Physical Health	–	Porteous et al., 2012 ; Ucci et al., 2011 **B,C,D,E,F**
9	*Airtightness*	*Lower air change rate*	*Changes in indoor air quality (IQA)*	*Increased exposure to indoor sourced pollutants. Decrease in external sources of pollutants (e.g. $PM_{2.5}$).*	*Physical Health*	*+/-*	*Wilkinson et al., 2009; Crump et al., 2011; Yu and Kim, 2011 B,C,D,E,F*

As illustrated in Table 2.3, some interventions have a cascade of consequences from their direct effects on the building, to effects on human wellbeing and the environment (nature). Columns B-D represents the flow of impacts caused by the application of airtightness policy on buildings. The resulting unintended consequences are seen in columns E and the domain affected in column F. Column G shows the direction of the impact; whether positive, negative or both. Column H shows the literature source and whether this refers to the whole flow or an aspect of it by indicating the columns to which the literature source refers. A full version of this table with all the further unintended consequences described in the included studies and additional references used can be found in Appendix B. A more complete consideration of the complex inter-relationship between airtightness and its unintended consequences is shown in Figure 2.2, illustrating the limitations of mapping each impact pathway in isolation when considering policy formation. The level of complexity seen raises a number of issues which are dealt with under the sections 2.2.4 *Summary of Impacts* and 2.4 *Discussion* below.

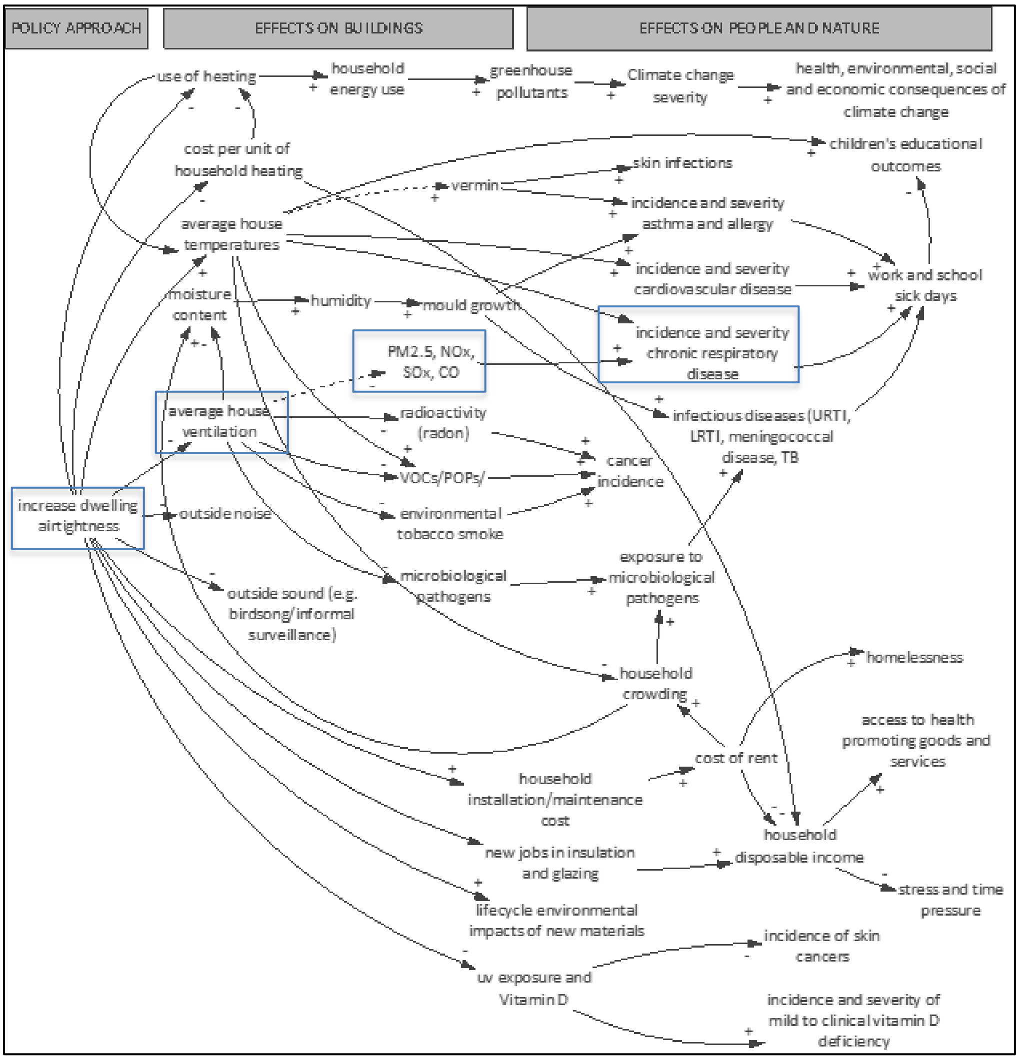

Figure 2.2 The complex links arising from the policy of promoting airtightness in the domestic stock and the impact on buildings, people and the wider environment. Arrows and lines denote known connections, +/- signs the direction of impact whether causing an increase (+) or a decrease (-) in impact.

2.2.2 Impacts Associated with Purpose Provided Ventilation

A key approach to dealing with the potential negative impacts of increasing the airtightness of dwellings, is to accompany these interventions with purpose provided ventilation (PPV) systems. However, a number of modelling studies reported that the addition of purpose provided ventilation to airtightness had its own wide ranging effects. Generally, a reduction in most indoor sourced airborne pollutants; $PM_{2.5}$, mould and environmental tobacco smoke (ETS) were reported, which yielded health benefits (Wilkinson et al., 2009; Milner et al., 2011). However, in practice many ventilation systems do not perform to their designed standards, with poor installation and maintenance cited as reasons for further reductions in capacity (Silva et al., 2012). Increased ventilation without heat recovery could lead to energy efficiency gains being offset by ventilation heat losses with GHG emission increased or remaining unchanged and increased fuel bills, especially so if systems are not understood by end users (Hesaraki et al., 2013; Mulligan and Broadway, 2010).

$PM_{2.5}$ levels can also be elevated indoors, especially when outdoor $PM_{2.5}$ levels are high. Certain filters, room air cleaners, and extraction equipment can help reduce indoor particle levels. They can also be reduced by not smoking inside, and by reducing usage of other particle sources such as candles, wood-burning stoves and fireplaces (Jones, et al., 2000; McMurry et al., 2004). In addition, increases in outdoor sources of pollutants (such as $PM_{2.5}$) within the indoor environment could occur if systems are not filtered or are not working correctly (Milner et al., 2011). The application of Mechanical Ventilation with Heat Recovery (MVHR) systems with filters, although proposed as a solution to these problems also has reported impacts, for example disturbed sleep resulting in systems being switched off (Balvers and Bogers, 2012). Poor installation and lack of maintenance of MVHR systems have also been linked to increases in indoor pollution and microbiological growth (Thorpe, 2011; Balvers and Bogers, 2012) and failure to achieve the energy savings anticipated from design data. On the other hand, studies have demonstrated that correctly functioning systems provide good air exchange and a quieter environment resulting in a reduction in household accidents and a general increase in mental alertness (Mendell and Heath, 2005). However, current MVHR systems may not be appropriate for the majority of existing properties requiring retrofitting due to the extensive duct work required (Sullivan et al., 2013).

2.2.3 Impacts Associated with Previous Models of Funding and Implementation of Policies

Implementation mechanisms and funding strategies influence the success of any policy. Effective marketing, the current economic uncertainty and loans for the green deal offered at higher interest rates than could be obtained elsewhere, are all issues that influence the success of policies to improve the energy efficiency of housing, with this key being cited as one of the main reasons for its demise. Cash back schemes offered as a means to encourage initial take up of energy efficiency products have been very limited when perhaps a subsidy on base material cost would be more effective (FMB, 2012). It would appear there is a reliance on voluntary public engagement 'altruism' which could lead to an increase in fuel poverty and the gap between the better-off and poor, with the neediest not benefiting from the policy (Davies and Oreszczyn, 2012; Rand Europe, 2012). If this is not addressed, policy failure might ultimately result in failure to curb GHG emissions from much of the existing housing stock (Dowson et al., 2012). The scope of finance offered is limited, such that with necessary façade and fabric repairs are excluded from the scheme, leaving the homeowner with additional costs for which they must source funding elsewhere (STBA, 2012). Damage to fabric and contents may occur if such a finance scheme is implemented, leading to possible failure to achieve the energy savings expected and possible issues with moisture ingress and health impacts (Historic Scotland, 2007; Davies and Oreszczyn, 2012). Additional costs needed may cause delays or a decision not to proceed with a scheme.

Holistic policies which tackle the issues of ventilation, indoor air quality (IAQ) and behaviour could help avoid multiple negative consequences from airborne pollutants such as $PM_{2.5}$ (Rand Europe, 2012) and impacts such as mould on building elements and contents (Kohler and Hassler, 2012). Schemes used to implement policies can have on-costs such as increased installation/maintenance costs, reducing disposable income and creating stress. In extreme circumstances this could lead to a "heat or eat" situation and a social determination of comfort (Hills, 2012; STBA, 2012). With current housing shortages, upgrades of dwellings in the rented sector could see increases in rents possibly resulting in overcrowding and increased exposure to airborne pollutants, pathogens, infectious diseases and could impact on social cohesion and mobility (Beggs et al., 2003; Noakes et al., 2006) This could have long term effects on future socio-economic wellbeing and status (Solari and Mare, 2012), negative impacts on child development (Leventhal and Newman, 2010); and additionally if rents become untenable; a risk of an increase in homelessness (Marmot, 2011).

Should future public uptake of schemes driven by energy efficiency policies prove successful, there are clear economic benefits led by the need for new designs, equipment, materials and specification with resulting economic growth, potential growth of UK based manufacturers, supply chains, specialist designers, contractors and general employment (DECC, 2014). However, as previously discussed, it is essential that this growth is sustainable and does not simply add to the carbon burden (Santarius, 2012). There is the opportunity for increasing the skill set of the current construction work force to ensure buildings reach specification (Pan and Garmston, 2012; Sinnott and Dyer, 2012) and increase partnership working (Latham, 1994; CTF, 1998) improving business prospects nationally and abroad.

2.2.4 Summary of Impacts

A summary of the downstream impacts on domestic properties caused by the application of the various energy efficiency measures investigated are shown in Table 2.4. In addition, the directions of the (unintended) consequences as seen in the literature search are shown. As previously noted, this table has been adapted from the framework in order to clarify specific impacts on domestic properties. A summary of the total impacted domains discovered are shown in Table 2.5, which illustrates how unintended consequences translate into impacts that affect people and their health, buildings, society and the environment, with many single consequences impacting multiple domains.

Table 2.4 Downstream impacts on buildings related to the application of the investigated energy efficiency measures and their direction of influence

Domain	Direction of influence			
	+ve	-ve	+/-ve	Totals
Physical health	16	47	13	76
Mental health	4	4		8
Psychological wellbeing	9	5	2	16
Child development	1	1		2
Social cohesion		3		3
Social inequalities		1		1
Social mobility		2		2
Occupant behaviour		1	2	3
Household finances		2	1	3
General economic	9	1	2	20
Building fabric	1	17	2	20
Legal		3		3
Environmental	7	31	9	47
Totals	**47**	**118**	**31**	**196**

Numbers refer to the number of references within the papers showing impacts on the particular domains

It should be noted that the totals seen in Tables 2.4 and 2.5 demonstrate where the attention of previous research has focused, rather than necessarily the relative importance of a particular influence on unintended consequences. However, it is apparent that physical health and the changes in indoor air quality (IAQ) (e.g. concentrations of $PM_{2.5}$) occupy a large proportion of research and this is considered an important subject for investigation. Table 2.5 highlights the individual routes to consequences for clarity and shows the range and domains impacted by policies to apply energy efficiency measures to the domestic stock. However, this method, although useful, hides the complexity and interconnections that exist between the different domains. Using the example of increased dwelling airtightness and its impacts on concentrations of $PM_{2.5}$ seen in Table 2.3 and Figure 2.2 shows that when taken together, the linkages identified in the literature form complex and dynamic inter-relationships between the individual components. This suggests the need for a complex and dynamic approach (such as micro environmental modelling) when investigating the impacts of changes in indoor concentrations of $PM_{2.5}$ brought about by the application of various energy efficiency measures on the housing stock.

Table 2.5 Domains of impact and their direction of influence

Downstream impacts on buildings	Direction of influence			
	+ve	-ve	+/-ve	Totals
Noise levels	4	4	2	10
Air change rates/Indoor air quality	9	6	9	24
Indoor temperatures and relative humidity	18	13	4	35
UVB, UV and UVA reception	2	9		11
Energy use		4	8	12
Fabric/Structural components	2	25		27
Totals	**35**	**61**	**23**	**119**

2.3 $PM_{2.5}$ and Health

In part, the emphasis on indoor air quality/air change rates and physical health seen in the literature is understandable given that changes in IAQ can have significant impacts on occupants. Particulate matter (PM) can affect human health. Research evidence suggests that it is the fine components, those less than 2.5 µm in diameter ($PM_{2.5}$), that are the main cause of the harmful effects of particulate matter, with a greater association between health effects and mass concentration seen as the particle size decreases (McGranaham and Murray, 2003). These fine particles come from a variety of natural and anthropogenic sources. Combustion sourced $PM_{2.5}$ include black carbon, trace metals and organic compounds. The health effects due to exposure to $PM_{2.5}$ are becoming well documented (McMurry et al., 2004; Peled et al., 2005; WHO, 2013a COMEAP, 2015;). The most severe effects are likely to be caused by exposure to particles over a long period of time. Health studies have shown a significant association between exposure to fine particles and mortality (Peled et al., 2005; WHO, 2013a). Other important effects include aggravation of respiratory tracts and cardiovascular disease (as indicated by increased hospital admissions, emergency visits-events associated with short term exposure events-absences from school or work and restricted activity), lung disease, decreased lung function, asthma attacks and certain cardiovascular problems such as heart attacks and irregular heartbeat and ultimately death (McGranaham and Murray, 2003; Peled et al., 2005, WHO, 2013b; Atkinson et al., 2014).

There does not appear to be the same level of evidence for short-term exposure i.e. pollutant spikes. Also, Pope et al., 2002 showed that the estimated increase in all-cause mortality effects of long-term exposure to PM_{10} (4-7% per 10 µg m-3) are far greater than those associated with daily exposure rates (1% per 10 µg m-3). Although PM_{10} contains the smaller fractions ($PM_{2.5}$); drawing conclusions from PM_{10} data can be problematic in distinguishing between impacts for the various fractions. However, more recent evidence has shown clear associations between $PM_{2.5}$ and adverse health impacts (; WHO, 2013b, COMEAP, 2015), although there are concerns regarding heterogeneity in effect estimates in different regions of the world (Atkinson et al., 2014). Regulations 23, 24 and 25 of the Ambient Air Quality Directive (2008/50/EC) introduce a new control framework for external $PM_{2.5}$. This has been formulated in response to the evidence of adverse health effects which indicate no safe threshold below which exposure would not pose a risk (WHO, 2013a). Consequently, health benefits are gained whenever levels of exposure to $PM_{2.5}$ are reduced. In the EU, average life expectancy is estimated to be 8.6 months lower as a result of exposure to $PM_{2.5}$ produced by anthropogenic activities (WHO, 2013a; DEFRA, 2009). In the UK exposure to $PM_{2.5}$ is a significant health issue (PHE, 2013). The UK fraction of mortality attributable to $PM_{2.5}$ is estimated to be 5.4% (based on outdoor $PM_{2.5}$ exposure), i.e. in excess of 24,000 deaths in 2011 (ONS, 2012).

Particles that are 10 µm or larger tend to be captured in the nose or in the tracheal and bronchial regions of the respiratory tract. Particles less than or equal to 2.5µm in aerodynamic diameter ($PM_{2.5}$) are referred to as "fine" particles and are believed to pose the greatest health risks (Englert, 2004). Because of their small size (approximately 1/30th the average width of a human hair downwards), fine particles can lodge deeply into the lungs and transfer across into the bloodstream. The deposits of particles in the lungs are not only influenced by particle size but also by concentration, composition, pH, and solubility (McMurry et al., 2004; Peled et al., 2005). Deposits will also vary among non-smokers, smokers and individuals with lung disease. Lung deposition is slightly higher in smokers and greatly increased in individuals with lung disease e.g. chronic obstructive pulmonary disease (COPD) or asthma (CDCP, 2010). Those most at risk include: young children, the elderly, people with existing lung diseases, those with jobs involving heavy physical exertion, people (particularly adults) exercising outdoors in polluted areas and smokers (WHO, 2016). Secondary effects include healthcare provision and loss of productivity (Sloss and Smith, 2000; Marcazzan et al., 2002)

Due to the direct impacts on health, and the associated costs, there is a strong need to understand and forecast the concentrations of $PM_{2.5}$ in the indoor domestic environment (as well as other environments) both currently and under a range of possible future scenarios. To achieve this there is a need to recognise all the factors contributing to increased exposure, such that effective public policy can be informed, and timely warnings can be issued to vulnerable groups (such as those with existing respiratory and cardiovascular conditions). In modelling the relative contributions of outdoor and indoor $PM_{2.5}$ sources should currently be modelled as separate components. The primary reason to distinguish between these $PM_{2.5}$ sources is the differences in the nature of the particles of indoor and outdoor origin (Adgate et al., 2007; Abdallah et al., 2013) and in their potential (but largely unquantified) relative toxicity, which are perceived as sufficiently different in magnitude as to require separate consideration (Long et al., 2001; Ebelt et al., 2005; Stanek et al., 2011; Rohr and Wyzga, 2012). This gives health impact assessments the opportunity to distinguish perceived relative risks to population health. However, this is an area of high debate as there are also potential differences in relative toxicity of outdoor particles from different sources e.g. combustion particles vs. sea salt (NPACT, 2013).

2.4 Discussion

This scoping, cross-disciplinary section of the literature review seeks to identify, enumerate and characterise what is already known about the broad range of consequences -some of which are unintended- of current interventions to reduce GHG emissions from the UK housing stock. Guided by a holistic framework (see Fig 2.1) for potential impacts more than one hundred consequences were discovered across a range of domains of human wellbeing, including physical, mental, social, environmental and economic wellbeing. It points to issues in IAQ, physical health (and specifically changes in $PM_{2.5}$) being a key consequence arising from the application of energy efficiency interventions.

For the examples outlined in detail, there are some individual targeted solutions suggested in the literature (Harlan and Ruddell, 2011; Oikonomou et al., 2012; STBA, 2012). However, there is always the danger that these single focus solutions are likely to have further unintended consequences. It has been demonstrated by the investigation of airtightness that when taken together, the linkages identified in the literature form complex inter-relationships between various domains, suggesting that more holistic, multi-disciplinary approaches are needed to formulating and implementing policies regarding housing. A similar approach is needed, when investigating individual consequences in detail, for example indoor $PM_{2.5}$ exposure as seen in this study.

The study of unintended consequences in the built environment, and indeed in other areas of society and policy, is as yet, underdeveloped. This is the first time that a holistic attempt has been made to characterise the effects of policies to reduce end-use housing energy demand. It builds on previous work aiming to integrate a range of physical and mental health impacts of policy options to reduce GHG emissions of the housing sector, significantly broadening the scope of impacts considered (Hamilton et al., 2012, 2015). This chapter is part of a larger project (and research paper) and is limited to an initial characterisation of consequences by the broad but non-systematic approach taken. It is not possible to draw conclusions about the size of intervention effects, or their relative importance, which will require further study. In addition, there are almost certainly likely to be a greater range of ‘unknown’ unintended consequences, which the current approach to research is not able to reveal and requires new methodologies to enable investigation (Davies and Oreszczyn, 2012).

However, some limited conclusions for policy can be drawn from this part of the review. Possible unintended consequences are related both to faulty policy formulation and to problems with implementation. In complex systems such as housing, policy formulation processes that focus on limited objectives (in this case GHG emissions reduction), while taking inadequate account of the complex and dynamic inter-relationships between objectives and outcomes, are vulnerable to policy failure and negative unintended consequences. On the other hand, a more integrated policy formulation process has the potential to achieve co-benefits across a range of objectives. This requires a different set of policy formulation methods that can bring a wide range of stakeholders together in a collaborative learning process about dynamic system complexity. Furthermore, it was clear from the review that choices relating to funding mechanisms for policies can either support or undermine policy objectives. Incorporating considerations about funding mechanisms into policy formulation could improve these choices. Investigation of changes in IAQ e.g. $PM_{2.5}$ in the context of unintended consequences of energy efficiency interventions has been shown to have received prominence as a leading issue in research, in part due to its known health impacts as shown above. However, it also suggests that in order to investigate the causes and influences on $PM_{2.5}$ concentrations in domestic properties (and elsewhere), a holistic framework is used and that any modelling captures the different stands of influence on $PM_{2.5}$ exposure.

2.5 Summary and Implications for Research

To avoid future policy failure and possible liabilities, there is an urgent need for processes that ensure regulatory measures are framed to achieve multiple realistic objectives, including those of high community priority. Part of this process will be the acceptance that multiple trade-offs (for example between GHG emissions reduction vs $PM_{2.5}$ and public health) will likely occur if policies are rigidly enforced as they stand.

As a major sector contributing to the UK's greenhouse gas (GHG) emissions, housing is an important focus of Government policies to mitigate climate change. Current policy promotes the application of a variety of energy efficiency measures to a diverse building stock, which will likely lead to a wide range of unintended consequences. This scoping review has identified more than 100 unintended consequences impacting building fabric, population health and the environment, thus highlighting the urgent need for Government and society to reconsider its approach. On this point, the current Government (as of June 2017) has, as previously stated, cancelled the Green Deal and other housing related policies and is in the process of formulating new policies, which will still need to help achieve the objectives of the Climate Change Act 2008. This thesis has assumed that this act will still be one of the primary drivers of new policy. As future policies are unclear, all modelling has to be carried out in the context of current building regulations and associated guidance.

Many of the impacts affecting $PM_{2.5}$ exposures are connected in complex relationships. Some are negative, others possibly co-benefits for other objectives. While there are likely to be unavoidable trade-offs between different domains affected (e.g. health) and the emissions reduction policy, a more integrated approach to decision making could ensure co-benefits are optimised, negative impacts reduced and trade-offs are dealt with explicitly. Integrative methods can capture this complexity and support a dynamic understanding of the effects of policies over time, bringing together different kinds of knowledge in an improved decision-making process and is likely to offer a useful route forward, supporting cross-sectorial policy optimisation that places reducing housing GHG emissions alongside other housing policy goals and could assist in such areas as changes in IAQ and more specifically indoor concentrations of $PM_{2.5}$, although the presence of other airborne pollutants may require an optimal strategy (Das et al., 2013)

This chapter has established the context of policy making that has led to a range of unintended consequences including the changes in IAQ and indoor domestic $PM_{2.5}$ concentrations in particular.

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Chapter 3

Factors Influencing Indoor $PM_{2.5}$ Exposure and Health

Introduction

In order to establish the range of factors influencing indoor domestic $PM_{2.5}$ exposure, so as to produce a robust methodology required to achieve the aims of this study, a review of current research literature is needed that encompasses the historical context of airborne pollution, the nature of $PM_{2.5}$ and the factors influencing indoor concentrations by identifying current data, theory, practice and areas of debate.

3.1 Search Methodology: Scope, Inclusion Criteria

The literature search process adopted was primarily thematic, with methodological elements included where appropriate. The scope of the literature review was defined and initial keywords established, which were subsequently used in literature searching programmes.

For example; in the case of indoor $PM_{2.5}$ sources, the initial keywords used were:

- Pollution indoor
- Emission rates $PM_{2.5}$
- Indoor emission rates $PM_{2.5}$ + sources
- Indoor air + quality + $PM_{2.5}$ + Aerosols
- Analysis + $PM_{2.5}$ + indoors

A comprehensive search was conducted using the following data bases: Web of Knowledge (including citation reports which were further investigated via Scopus); Google Scholar; Index of Theses; Science Direct; Social Science Research Network and PubMed. 'Grey' literature investigated included the Open Grey data base, European Union and UK Government legislative and policy documents, technical data sheets and specifications, published textbooks, reports from NGO's involved in the retrofitting process, recognised websites (for example from construction organisations) and other web-based articles. This grey literature was used to identify further peer-reviewed studies. Additional terms and combinations revealed by the literature search were also investigated. The search was limited to relevant studies that were published in the English language from 1970-2017. Those accepted were placed in *Mendeley*, a resource management software package. This enabled a matrix to be created that illustrated further keywords used by authors in the field and references, enabling a more complete search to be carried out.

3.2 $PM_{2.5}$: Historical Background and Context

Urban air pollution, and specifically particulate matter is not simply a modern phenomenon. Seneca, Emperor Nero's tutor, who suffered ill health (thought to be lung related), wrote in AD 61 that he felt better once he had left 'Rome's oppressive fumes' (Brimblecombe, 1987). Additionally, the Roman courts had to deal with cases arising from complaints of smoke particles from factories penetrating the homes of its citizens (O'Riordan, 1995). The first documented case of a domestic indoor air pollution incident in Britain occurred in 1257 when the wife of Henry III left Nottingham Castle in fear for her health due to the 'stench' of burning sea-coal smoke - a mixture of sulphur and PM (Brimblecombe in Elsom, 1992). In 1273, Edward 1 tried to ban the burning of coal in London to deal with the problem of pollution by PM in the form of smoke particles (Brimblecombe, 1987). The Industrial Revolution (c. 1750 onwards) compounded the problem with its high use of fossil fuels; primarily coal (Miller, 1996). In 1897, the meteorologists Mossman and Brodie identified the first link involving pollution and the formation of atmospheric effects such as the London Smogs (a mixture of fog and particles of soot), by assembling a 200-year data set from registers and diaries and a 20-year data set from official records between 1870 and 1890 (Brimblecombe, 1987; Robinson and Henderson-Sellers, 1999).

Since the industrial revolution, rapid urban and industrial growth coupled with intensive agricultural practices and the rise of new technologies (particularly the combustion engine) has caused vast quantities of potentially harmful waste products (including $PM_{2.5}$ and its gaseous precursors) to be released into the atmosphere at rates which exceed assimilative capacities, often without a clear understanding of the environmental repercussions at the time, or a reluctance to accept the consequences (Daly, in Dresner, 2002). This airborne pollution has affected the health and wellbeing of the population, causing damage to crops, vegetation and wildlife, as well as building structures (McGranaham and Murray, 2003; McMurry et al., 2004). It has altered the climate on a global scale, resulting in the degradation and depletion of the very natural resources (Earth capital) needed for long-term sustainable development (Elsom, 1992; Kerry-Turner et al., 1994; Dresner, 2002).

3.3 $PM_{2.5}$ Characteristics

Particulate Matter (PM) is a major airborne pollutant form. Airborne PM includes a wide range of particle sizes and many different chemical constituents. Airborne particles can range in size from a few nanometres (nm) to around 100 micrometres (μm) in diameter. They can be solid or liquid and may contain an internal mix of species and phases. They may be spherical or irregularly shaped and the surface composition may be different to the bulk composition (McMurry et al., 2004). The greatest number of particles falls into the ultra-fine size range (less than 100 nm), whereas although the larger particles contribute little to particle number, they often represent the greatest proportion of particle mass (McMurry et al., 2004). $PM_{2.5}$ refers to particulate matter which passes through a size-selective inlet as defined in the reference method for the sampling and measurement of $PM_{2.5}$, BS EN 14907: 2005, with a 50% efficiency cut-off at 2.5μm aerodynamic diameter. As such, the term '$PM_{2.5}$' encompasses all material less than or equal to 2.5μm in size.

PM can be divided into primary and secondary components, the former produced directly by various industrial processes and combustion emitted directly into the atmosphere; the latter being formed within the atmosphere as a direct result of chemical reactions of gaseous precursors such as NOx and SOx, often with the assistance of sunlight that forms compounds that can condense, coalesce or collide to produce particulate matter (PM). However, droplet-based PM may also form as a result of nucleation or coagulation and can undergo exchanges with other atmospheric organic compounds resulting in phase changes from gas to condensate and back again (McGranaham and Murray, 2003; McMurry et al., 2004). The formation of secondary particulate matter may take hours or days. During this time the air containing the pollution can travel long distances (McGranaham and Murray, 2003). Most gaseous precursors and $PM_{2.5}$ have the potential to travel continentally > 10^6 m (McMurry et al., 2004), although the distance travelled is related to the source and input point such that the predominant wind system has an influence. Both particle size and mass affect the ability to be entrained in airflow.

$PM_{2.5}$ sourced internally in domestic properties, tends to have different chemical compositions than those sourced outside, enabling source apportionment based on characterisation and a clear indication of indoor outdoor (I/O) ratios (Gehin et al., 2008). Indoor sources of $PM_{2.5}$ have been linked to materials used in building construction; fixtures and fittings; appliances and fuels used for cooking and heating, as well as smoking and a variety of human domestic activities (Weschler, 2009; Meng, et al., 2009). Outdoor $PM_{2.5}$ comes from a variety of sources both natural and anthropogenic. Principle sources are combustion (e.g. motor vehicles, power stations and domestic heating), mechanical activity (e.g. quarrying and agricultural harvesting) and natural processes (e.g. entrainment of soil by the wind, generation of marine aerosol particles, forest fires, volcanic activity and biological material including spores) (Jones et al., 2000; McMurry et al., 2004; Witham and Manning, 2007).

3.4 $PM_{2.5}$: Outdoor Sources, Sinks and Measurement

As previously stated, outdoor particulate matter comes from a variety of sources. Primary outdoor sources are combustion based (motor vehicles, heating and power stations), mechanical processes (e.g. quarrying, construction and agricultural activities) and natural processes (e.g. entrainment of soil by the wind, generation of marine aerosol particles, volcanic activity, and biological materials). Forest fires along with space heating of buildings, plus high levels of spores in season can contribute to an increase in particulate matter and represent the balance of sources (Jones *et al*., 2000; McMurry *et al*., 2004; Witham and Manning, 2007). Although the overall distribution between PM_{10} and $PM_{2.5}$ is dependent on source, location and secondary reactions, there is generally a bimodal distribution in the ambient air Figure 3.1 (McGranaham and Murray, 2003).

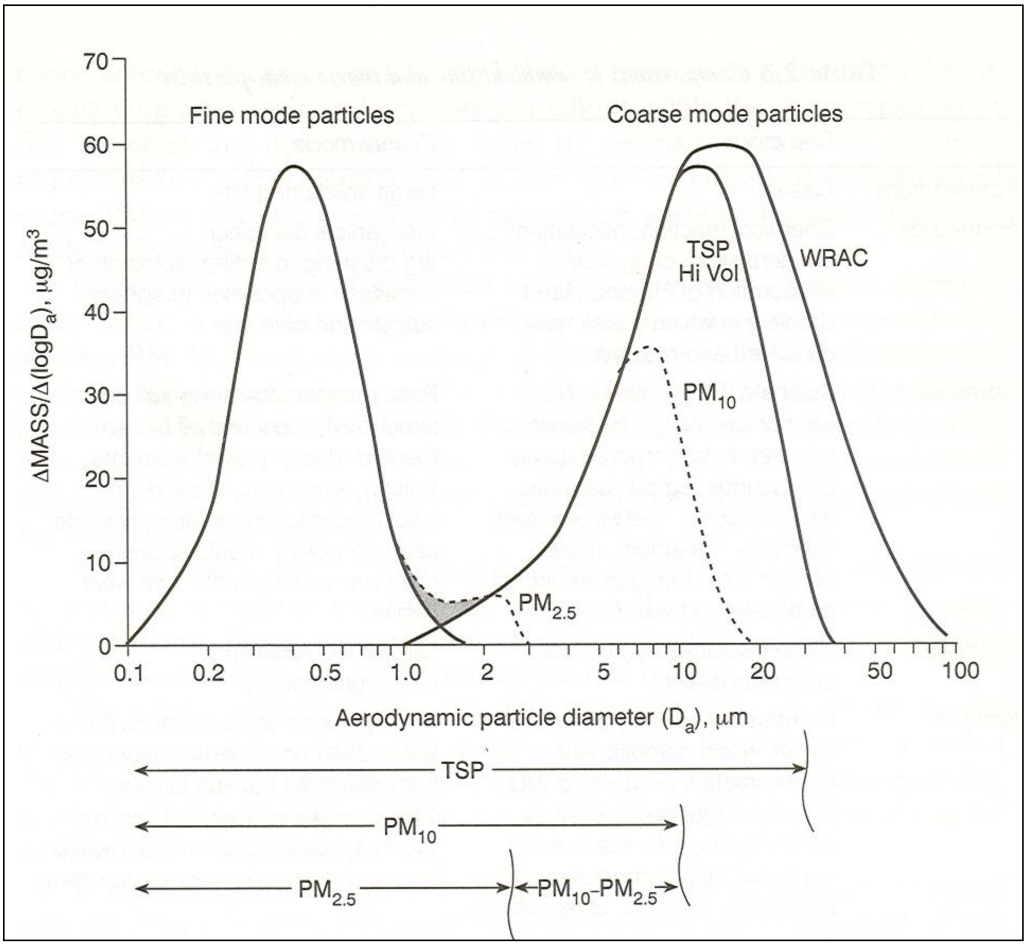

Figure 3.1 Representative example of a mass distribution of ambient PM as a function of aerodynamic particle diameter. TSP is the total suspended particulate matter. WRAC: wide-ranging aerosol classifier, which provides an estimate of the full coarse mode distribution (*Source:* McGranaham and Murray, 2003).

3.4.1 Primary Outdoor Components of $PM_{2.5}$

There are a range of materials which can contribute to the primary components in the PM category (Robinson and Henderson-Sellers, 1999; Park, 2001; McMurry *et al*., 2004). These include:

- **Sodium Chloride** sea salt
- **Elemental Carbon** black carbon (soot) is formed during high temperature combustion of fossil fuels and biomass fuels.
- **Trace Metals:** these metals are present at very low concentrations and include lead, cadmium, nickel, chromium, zinc and manganese. They are generated by metallurgical processes (e.g. steel making); from impurities in fuel additives and from mechanical abrasion processes (e.g. brake and tyre wear on vehicles).
- **Minerals** these minerals are found in coarse dusts from quarrying, construction and demolition work, via wind driven dust. They include compounds of aluminum, silicon, iron and calcium.

There are both natural and anthropogenic sources of primary atmospheric $PM_{2.5}$. The largest natural sources are windblown dust, volcanoes and material from forest fires. Sea spray contributes a large source but invariably, most falls back to the ocean or close to where it is emitted (Robinson and Henderson-Sellers, 1999; Park, 2001).

In most urban sites elemental carbon represents the largest man-made source via fossil fuel combustion. Urban sites, such as the Greater London Authority (GLA), have many years of data for statutory pollutant monitoring stations, that show mean $PM_{2.5}$ concentrations can range from 4-8 times the estimated natural background levels (LAQN, 2010). This clearly implies that anthropogenic activities including transport make substantial contributions to $PM_{2.5}$ loadings in urban conditions (McMurry et al., 2004). In local situations up-wind from larger industrial sources, or where these are not present, transport is likely to be the primary contributing sector. The overall composition of $PM_{2.5}$ at a point in time can vary greatly with pollutant levels. Differing source contributions and metrological conditions usually result in a range of both seasonal and diurnal variations in both $PM_{2.5}$ mass concentration and composition (McMurry et al., 2004; Witham and Manning, 2007).

Secondary particles form in the atmosphere resulting from chemical reactions that lead to the formation of substances of low volatility from gaseous precursors, which consequently condense into the solid or liquid phase, thereby becoming particles. Particle size and mass affect the ability to be entrained in airflow. Most gaseous precursors and PM_{10} can potentially travel regionally between 10^4-10^6 m, with the lighter fractions, $PM_{2.5}$ travelling continentally > 10^6 m (McMurry et al., 2004). Biomass fires in western Russia, in May 2006, caused long range transport of PM which registered elevated levels on the UK automatic urban and rural air quality monitoring network (AURN), with an hourly maximum of 163 µg m^{-3} in Scotland and readings > 130 µg m^{-3} recorded at seven urban sites in England and Wales between 8-10 May 2006 (Witham and Manning, 2007). This infers the importance of both local and synoptic wind patterns in pollution formation and distribution. Pollution rose analysis indicates that under south-easterly winds many areas of the UK experience an increase in mean airborne $PM_{2.5}$ concentrations of up to 30% over the average for all directions. This is mainly attributable to long range transport, although a lack of boundary layer ventilation is suggested as an additional contributing influence (Rigby et al., 2006).

3.4.2 Vehicle Traffic as a Source of $PM_{2.5}$

Road transport is a major source of airborne pollution with over an estimated 21% of UK domestic greenhouse gas emissions in UK are sourced via this sector and up to 23% of total $PM_{2.5}$ emissions (DfT, 2015). However, individual pollutant contributions can be higher, for example an estimated 95% of carbon monoxide (CO) in North American cities comes from road vehicles (McMurry et al., 2004). Locational factors also play a part, with little polluting industry; the contribution of transport can be the primary source of $PM_{2.5}$ (Bullard et al., 2000). This may be the case for example in the Greater London

Authority area (GLA) as the site is bisected by numerous major arterial routes, but is within an urban area with a very high residential component and little polluting industry.

Alternative energy forms, such as hybrid electric and hydrogen fuel cell vehicles, although promising technologies, are still years away from widespread use. Catalytic converters and cleaner fuels have substantially reduced the quantity of pollutants from some vehicles, however, they cannot reduce $PM_{2.5}$ over the entire size range (Mohr et al., 2006). Tighter emission standards for vehicles have been formulated as these improved engine technologies and fuels have become available, with the MOT test intended to identify vehicles that fail to meet the standards, thereby requiring repairing or scrapping. Vehicle exhaust testing has been included in the MOT test since 1991. This is especially pertinent as badly maintained/older cars can produce up to 40 times the amount of pollution as a cleaner car including increases in $PM_{2.5}$ emissions (McMurry et al., 2004; DfT, 2008, DEFRA, 2012).

However, this reduction in emissions has been underwritten by a general increase in car ownership. The GLA and the rest of the UK has seen a substantial increase in both car ownership and traffic flows. The 2011 census of private car and van ownership alone indicates an increase of 3.4 million (14%) between 2001-2011 (ONS, 2012) This is in part reflective of travel needs that are not being satisfied by other means of travel such as public transport, cycling and walking and may be a reflection of cost and reliability as well as perceived road safety and quality issues (Elsom, 1996). By the end of 2010 car ownership in the UK stood at 28.4 million and is forecast to increase by 38.8% between 2010 and 2040, to around 39.4 million cars in 2031 (DfT, 2013. Euro emissions standards (in particular Regulation (EU) 2016/427) sets tighter vehicle emission standards (EU, 2016). However, in terms of $PM_{2.5}$ it only covers exhaust PM emissions and PM originating from vehicles tyre and break wear are currently not part of vehicle regulations. Without any further technological improvements or increases in electric vehicles, any overall decrease in $PM_{2.5}$ emissions may be counterbalanced by the increase in numbers of vehicles. If other sectors, for example industrial and the power sectors come under tighter environmental legislation, as seems likely then, that the overall contribution of traffic as a source of $PM_{2.5}$ may remain static. In addition, there is uncertainty in light of the recent decision to leave the EU, as to what will be the standard for the UK?

3.4.3 The Urban Heat Island, Urban Air Circulation and $PM_{2.5}$ Movement

Urban areas, for example the Greater London Area (GLA) are frequently a few degrees warmer than the surrounding suburbs and countryside. The difference is related to building density, quantity of incorporated vegetation, over-paving, the heat absorption characteristics of the built environment and the artificial generation of additional heat sources within the city such as industrial processes, domestic heating and traffic flow (Strauss and Mainwaring, 1991; Robinson and Henderson-Sellers, 1999). In calm or near-calm atmospheric conditions this can result in a city generating its own atmospheric circulation which will impact the movement and concentration of $PM_{2.5}$ both generated within the urban area and transported from the outskirts. To observe such effects and as part of the DAPPLE project, Martin et.al 2008, released tracer gases in the urban environment and observed the horizontal air movements, direction and wind speeds; whilst Patra et al. 2008 carried out on-street observations of $PM_{2.5}$ movement and dispersion due to traffic on urban roads. Warmer air rises in the city centre and then moves aloft to the outskirts, while cooler air is drawn in from the suburbs and rural fringe by horizontal advection comparable to a sea breeze effect (Heaviside et al., 2015). Katabatic effects may strengthen these breezes if the urban area lies in a valley or basin, as on cooling, the denser air tends to flow downhill. This may influence other elements of the climate (such as convection and cloud formation) by providing a hot spot, leading to an increase in cloudiness over and immediately downwind of the city (Elsom, 1992; Woodcock, 1994). In addition, the regional airflow may be disrupted by man-made barriers creating instabilities in the airflow, this forces air to rise, increasing the possibility of cloud formation (Robinson and Henderson-Sellers, 1999).

In a turbulent atmospheric situation, the variety of shapes, structures and materials in an urban environment produce a rougher surface than the surrounding rural areas. At the street level this causes additional stresses on structures. The airflow around a particular obstacle is dependent on flow characteristics, the geometry of the building and the surroundings. This increases the leeward eddies in turbulent conditions (Barry and Chorley, 1998). Wake vortices occur and a stream of these, known as a Karmen vortex street (Figure 3.2), can travel downstream of the obstacle, causing a distinct pressure drop, compared to the surrounding wind. Venturi effects (Figure 3.3a) can cause increased wind speed and pressure where buildings converge and air is channelled between them, alternatively transverse currents can occur in complex structural layouts (Figure 3.3b) (Robinson and Henderson-Sellers, 1999; Park, 2001). Each new construction or the demolition of an existing structure affects the airflow patterns, but can be theoretically predicted using CFD techniques due to the complex physics involved (Vardoulakis et al, 2011). However, this is only likely to be carried out for individual buildings due to the processing power/time required to run effective simulations, although this is improving. Wind tunnel simulations show certain generalised characteristics such as flow separation and acceleration around obstacles (Robinson and Henderson-Sellers, 1999). All of these factors dictate the movement of airborne pollutants including $PM_{2.5}$ within the urban environment, creating temporary localised concentrations or dilutions, relative to the pollutant source and general urban background level of $PM_{2.5}$.

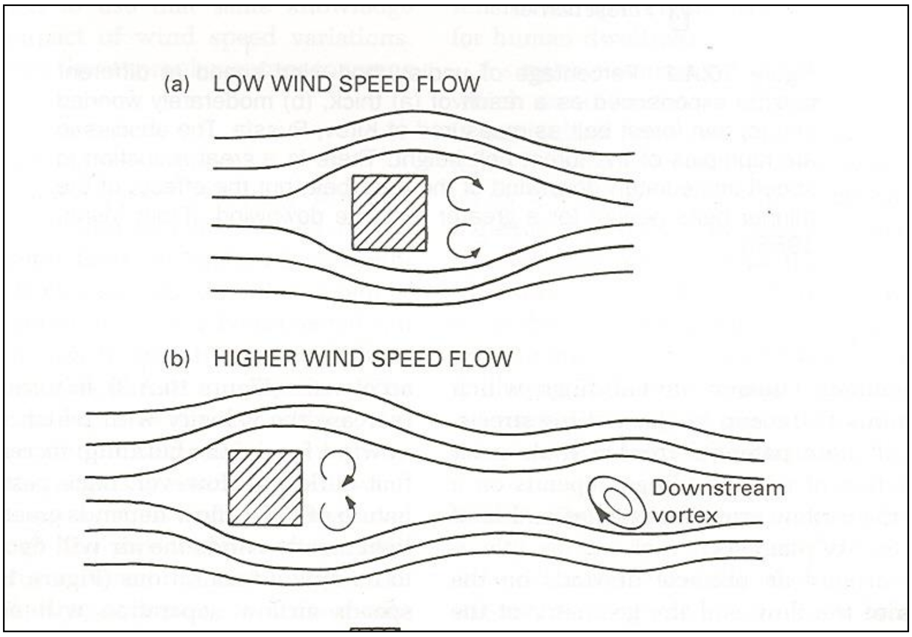

Figure 3.2 (a) Schematic representation of airflow around an obstacle. At higher wind speeds (b), downstream vortices can be created (*Source:* Munn, 1966 in Robinson and Henderson-Sellers, 1999).

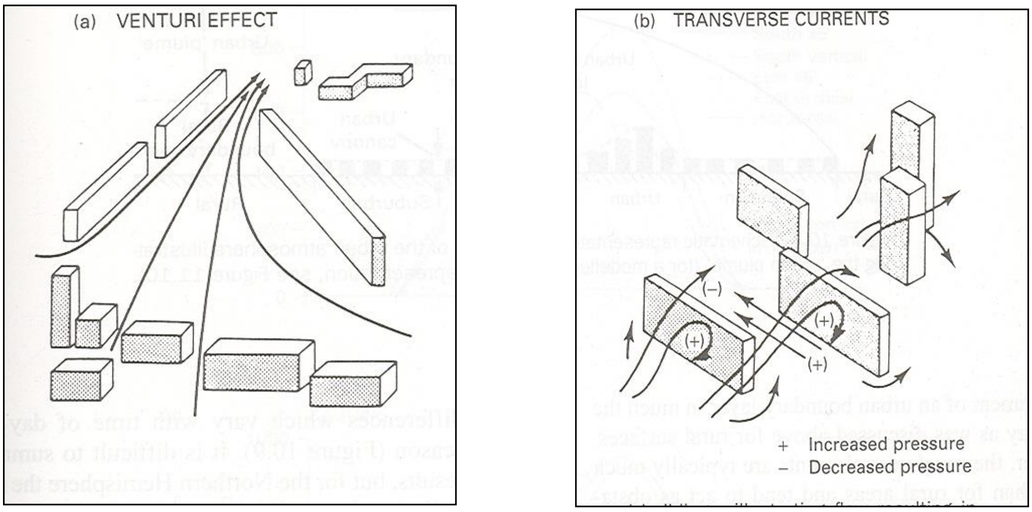

Figure 3.3 Two typical airflow patterns around buildings, illustrating flow resulting in (a) the Venturi effect and (b) transverse currents (*Source:* After Thurow, 1983 in Robinson and Henderson-Sellers, 1999).

3.4.4 External $PM_{2.5}$ Concentration and Dispersal

The height of the atmospheric boundary layer (ABL) effectively impedes the upward movement of particulate matter. When this is low, there is a reduced volume of air available to mix pollutants and so higher $PM_{2.5}$ concentrations are possible, especially under higher atmospheric pressure static systems. The height of the ABL varies with local climatic effects; with still conditions (such as a winter night), the ABL will be at its lowest. Conversely, in midday summer temperatures with high convection rates, the ABL will be higher thereby diluting the concentration for a given input of $PM_{2.5}$. Diurnal factors also affect the ABL height (Barry and Chorley, 1998). Surface temperature inversions play a major role in air quality, especially during the winter when these inversions are the strongest. A warm air layer above a cooler air layer acts like a lid, suppressing vertical mixing and trapping the cooler air at the surface. As $PM_{2.5}$ from vehicles, heating and industry are emitted into the air; the inversion traps these pollutants near the ground, leading to increased concentrations and poor air quality (Robinson and Henderson-Sellers, 1999; Kassomenos et al., 2014).

Rigby et al. (2006) found that under easterly and south-easterly winds, many areas of the UK including the GLA, experience an increase in particulate matter concentration of up to 30%. An unusually high number of easterly and south-easterly winds in February and March 1996 resulted in an increase in the number exceeding the PM_{10} 24 hr average target across the UK monitoring network, which included an increase in the $PM_{2.5}$ component (AQS2, 2007). Although previously attributed solely to long-range transport of particulate matter from Europe, it coincided with a 45-55% reduction in wind speeds as the warm continental air stabilised the boundary layer and reduced its height, thereby further increasing pollution concentration (Rigby et al., 2006). In Spring 2014, anticyclonic conditions lead to the 'Saharan dust' episode, when the leading edge of a storm collected and deposited material (within the PM size range) over southern UK, substantially increasing pollutant loadings (Macintyre et al., 2014). However, Vieno et al. 2016 showed that the elevated PM during this period was mainly driven by ammonium nitrate, much of which was derived from emissions outside the UK and that the Saharan dust only had an impact in the latter stages of the episode. Persistent anticyclonic (high pressure) conditions generally coincide with periods of heavy pollution concentration by stabilising atmospheric conditions, allowing only weak surface movements of air masses (Boix et al., 1995). Frioud et al., 2003 observed diurnal patterns in the distribution and concentration of particulate matter that follows the thermal wind cycle and the temperature inversions, under strong anticyclonic conditions in the Rhine Valley. A study in the Salt Lake City of North America showed only minor influences from background synoptic winds, with the major influences being thermally-driven local wind systems constrained by topography, leading to clearly defined diurnal cycles of $PM_{2.5}$ concentration (Alexandrova et al., 2003). Temperature inversions can occur as a result of a cold frontal invasion that pushes the warm air above it, or in an urban situations pollutants rising under convection are heated in the lower atmosphere, stabilising the atmosphere and creating a lid over the area that results in increases in $PM_{2.5}$ and other pollutant concentrations (Frioud et al., 2003).

Osborn and Jones (2004), discuss the factors controlling local climate in urban situations, from external forcing to site characteristics that dictate the local microclimate. Topographic influences can be important at the local level, producing changes in the overlying wind speed and direction. Local climate will however, still be subject to seasonal patterns such as increased wind speeds in winter months and diurnal temperature changes (Osborn and Jones, 2004). A study within the Castellón region of Spain showed a decrease in $PM_{2.5}$ concentrations during the winter months which were attributed to stronger winter winds, however this was dependent on the wind direction. High winter temperatures resulted in greater convection and a further decrease in pollution concentration where no temperature inversion occurred, though no such correlation was observed in the summer months. Lower concentrations were observed, when strong breezes dispersed pollutants. In addition to diurnal changes, a clear marked seasonal cycle was noted in the concentration of particulate matter (Boix *et al.*, 1995). A further study in the Kathmandu valley of Nepal showed monthly variations of $PM_{2.5}$ concentrations between rural and urban areas, regardless of season. Both the $PM_{2.5}$ concentrations and the degree of air mixing were closely associated with the temperature, wind speed and direction, along with strong diurnal variations in wind speed and temperature (Aryal et al., 2008). Concentrations of $PM_{2.5}$ in the urban German town of Trier were strongly affected by the surrounding Moselle Valley, with the overlying wind direction altered by relief dictating the $PM_{2.5}$ levels in the main urban area. Vertical mixing was constrained by temperature inversions in winter (Junk et al., 2003). Although topographic effects in the GLA are far subtle than those seen in the Moselle Valley; its position due to its basin location will influence air movements and therefore airborne pollution transfer.

Temperature inversions in urban sites such as the GLA (and/or urban heat island flow) may have the effect of temporarily trapping pollutants, thereby increasing $PM_{2.5}$ concentrations. The incidence of particular wind direction in urban situations can lead to changes in particulate concentrations (Rigby *et al.*, 2006). However, in general, high winds may have the effects of dispersing or diluting PM and low winds may assist in PM accumulation. There is a distinction between PM_{10} and $PM_{2.5,}$ in that $PM_{2.5}$ concentrations will be diluted, but heavier PM_{10} may be re-suspended thereby increasing the concentration of PM_{10} component previously subject to dry deposition (Kim et al., 2005; AQS2, 2007). Experimental studies of wind directional variability have shown that urban topography can have a significant effect in altering the overlying synoptic wind flow (both in terms of wind speed and direction) and therefore the persistence or dispersal of pollutants, leading to temporary localised differences in pollutant concentrations (Bullard et al., 2000).

At a smaller scale such as an urban street, a within-canyon process often occurs and dispersion, plume activity and vortex dispersion is greatly influenced by both the emissions from vehicles and other sources, along with actual traffic flow itself (Micallef and Colls, 1999; McGranaham and Murray, 2003). Using generalised addictive modelling to measure the relative importance of meteorological factors and traffic volume to pollution levels in an urban road, Aldrin and Hobeak Haff, (2005) concluded that the most important variables are traffic volume, wind direction and speed. Consequently, vertical and horizontal gradients of $PM_{2.5}$ occur within the urban canopy (Martin et al., 2008; Patra et al., 2008. Other predictor variables such as temperature and precipitation are not as critical, although $PM_{2.5}$ concentration is negatively associated with rainfall as PM acts as condensation nuclei. In order for the model to function, data on predictor variables of wind direction, wind speed, temperature, time of year and day, and traffic volume were required (Aldrin and Hobeak Haff, 2005).

3.4.5 $PM_{2.5}$ Outdoor Removal

Externally, various factors including building location, height and orientation to outdoor pollutant source and meteorology affect outdoor $PM_{2.5}$ contributions to indoor concentrations (Godish and Spengler, 2004; Patra et al., 2008). Removal of atmospheric $PM_{2.5}$ occurs in various ways. $PM_{2.5}$ can act as condensation nuclei becoming entrained in cloud formation processes and subsequently removed in precipitation. Alternatively, they can be physically removed by collision with raindrops (both processes referred to as wet deposition), with increasing precipitation inducing lower concentrations of particulate matter (Aldrin and Hobeak Haff, 2005). Dry deposition occurs as the pollutants come into contact with the ground or adjacent surfaces, though these can be re-suspended. Long or short-term transport of particulate matter to other locations is also possible (Elsom, 1996; McGranaham and Murray, 2003). $PM_{2.5}$ can also be transferred indoors and there is a relationship between external concentrations and internal $PM_{2.5}$ that is primarily dependant on location to pollutant source, local metrological effects and building permeability (Ozkaynak, et al., 1996; Milner, *et al*., 2005; Meng et al., 2009). Increasing wind speeds will generally disperse $PM_{2.5}$ but also re-suspend some particles, but generally reduce ambient $PM_{2.5}$ concentrations.

3.4.6 External $PM_{2.5}$ Measurement and the UK Monitoring Network

External monitoring is mandatory as a result of a statutory duty imposed under the Ambient Air Quality Directive (2008/50/EC) and requires large and complex equipment that can be left to monitor continuously for long periods. This equipment is able provide hourly, daily, and annual $PM_{2.5}$ concentrations, measured in µg m-3 (LAQN, 2010). One of the key elements in the development of air pollution control in Britain, was the realisation of the need for a national network of pollution monitoring stations. In the 1950s and 1960s this process began with stations measuring Particulate Matter and SO_2, seen as the primary pollutants, in response to the Clean Air Act of 1956 (Brimblecombe, 1987; Elsom, 1992). Automatic monitoring began in the 1970s with the arrival of new technologies and the formation of the UK Automatic Urban and Rural Network (AURN). This expanded from 30 sites in 1993 to around 300 in 2016. These additional sites were added to measure other pollutants including PAHs, NO_2 and SO_2. However, at the end of 2005, the national automatic NO_2 network was closed in part as local authorities had been given the duty to review and assess air quality in their areas under the Local Air Quality Management (LAQM) under Part IV of the Environment Act 1995. Many local authorities and boroughs continued to monitor for PMs and other pollutants via their own monitoring stations and also monitored NO_2 using a network of passive tubes. These systems run concurrent to national monitoring (DEFRA, 2008; UKAQA, 2008).

Local authority pollution monitoring stations generally use ambient particulate monitors. These instruments incorporate a true micro-weighing technology (developed and patented by Rupprecht and Patashnick), called a Tapered Element Oscillating Microbalance (TEOM). The hardware used can be configured to measure PM_{10}, $PM_{2.5}$ or total suspended particles (TSP) concentration. TSP is rarely used as it was found to be defined by the size-selectivity of the inlet to the filter. The size cut, which varied with wind speed and direction, was from 20 to 50 µm in aerodynamic diameter. Under windy conditions the mass tended to be dominated by large wind-blown soil particles of relatively low toxicity (RPCO, 2009). The TEOM was until recently, the only sampler used on the UK Automatic Urban and Rural Network (AURN) for national monitoring and was the only direct monitor in which the output was directly related to particle mass. The principle guiding its operation is that the frequency of mechanical oscillation of an element such as a glass tube; is directly proportional to the mass of the tube. Consequently, any changes in the effective mass of the tube resulting from the deposition of PM on the surface of a filter at the free end of the tube, result in a subtle but detectable change in its resonant frequency (Green and Fuller, 2006). The national particle standard has a principle of a 24-hour running mean in its concentration levels. The TEOM was one of only a few PM monitors which were able to provide data that could be summed over any consecutive 24-hour period. As a result, it became the natural choice of the then Department of the Environment AURN network (Green and Fuller, 2006).

There have been a number of issues regarding the integrity of TEOMs to monitor $PM_{2.5}$. Initially, condensation and the formation of water droplets caused reading errors. The addition of a heating element keeping the instrument at a constant 50ºC solved this. However, this resulted in failure to capture some particles due to evaporation of volatiles (Stedman et al., 2007). Research regarding inaccuracies in the use of TEOM monitors to measure $PM_{2.5}$, has shown an averaged underestimation of 51 + 24% during winter and of 35 + 26% during summer (based on µg m-3 hourly mean data) (Eatough et al., 2003; Favez et al., 2007). Set inlet size is critical, as the TEOM can only be adjusted to collect one group of PM and is not capable of distinguishing the quantity of smaller fractions of PM such as PM2.5 and PM1 when the inlet is set to receive PM10 (Sloss and Smith, 2000). The software configuration can dictate the accuracy of PM measurements and needs to be consistent across the network, as is now the case with one system in place (Green and Fuller, 2006).

DEFRA and the devolved administrations undertook a detailed study (2004-2006) on the equivalence of various sampler and instruments in terms of measuring $PM_{2.5}$. It concluded that the TEOM failed to meet the Data Quality Objective for overall uncertainty of 25%, as defined within the Ambient Air Quality Directive (2008/50/EC) (UWE, 2008). As the UK national networks are largely founded on the use of the TEOM analyser, a default correction factor of 1.3 is currently applied to the TEOM data, in order to provide a gravimetric-equivalent result that attempts to compensate for the loss of the volatile component of $PM_{2.5}$. One outcome of the study is that the TEOM analyser cannot be considered equivalent to the European reference method within the UK, even if a 1.3 slope correction factor (or any other factor) is applied (DEFRA, 2008). Of the monitors that meet the equivalence criteria, including the OPSIS SM200, Partisol 2050 and BAM; the Filter Dynamic measurement systems (FDMS) performed best in the tests. Since this, all the TEOM Monitors on The AURN network have been removed and replaced with FDMS monitors (UKAQ, 2010) and it is data from these monitors that are used in this study.

The FDMS uses microbalance technology to measure both the core mass and volatile fractions by measuring the positive and negative mass effects on the filter, of volatiles as they occur. This provides a more accurate and true measurement of total airborne particulate matter concentration, or a particular fraction such as $PM_{2.5}$ depending on the inlet setting (UKAQA, 2010). The more recent upgrade of equipment from TEOM to FDMS monitors (in many case both run concurrently) has enabled the AURN network to fully comply with the monitoring requirements within the new Ambient Air Quality Directive (2008/50/EC) which was transposed into national legislation on the 11th June 2010. Annual data sets are available from June 2011 which yield overall seasonal and annual data, enabling quantification of annual average external ambient $PM_{2.5}$ concentrations (UKAQ, 2010). The failure of the UK to meet the EU targets has led to increasing pressure from the EU in terms of potential fines as well as pressure from UK NGO's such as Client Earth, who have successfully taken the government to court over the issue and won.

3.4.7 Current Methods of External $PM_{2.5}$ Modelling

Traditional air quality monitoring stations only provide site specific measurements for $PM_{2.5}$, although urban background stations are thought to be indicative of the general level of pollution in a GLA borough (LAQN, 2010). Consequently, analysis of multiple urban background stations in a given area can yield average annual $PM_{2.5}$ concentrations for input into building physics models for indoor $PM_{2.5}$ concentrations of pollution movement and from there into health impacts assessment software, as health data relies on annual averages for calculation of changes in relative risk to $PM_{2.5}$ exposure (Wilkinson et al., 2009; Hamilton et al., 2015). Calculation of average annual $PM_{2.5}$ for the GLA are concentrations covered in Appendix D.

Wind roses, based on compass points, were developed by meteorologists to show the distribution of wind direction experienced at a given location over a considerable period of time, usually a year (Boix et al., 1995). These have since been adapted to produce pollution roses, which show the mean concentration of a particular airborne pollutant type in relation to its originating direction over time (EPA, 2008; Met Office, 2008). Pollution concentration, wind speed and directional data are available, but access to data regarding boundary layer height is generally not. As a result, interpretation in terms of point source attribution of $PM_{2.5}$ may not be possible or at best incomplete. However, it may be possible to give indications of the predominant direction of pollutant flow which can be related to wind direction, as a result of local effects (Rigby et al., 2006).

General dispersion modelling can be a valuable tool in the air quality assessment processes. Pollutant emissions arising from different source types (for example, $PM_{2.5}$ from road traffic); can be taken into account in terms of their impact upon ground-level concentrations (Stedman et al., 2007). These can be predicted across a wide geographical area and can assist with the determination of the geographic boundaries of any exceeding of air quality objectives. This can also indicate locations where outdoor $PM_{2.5}$ influences on indoor concentrations are likely to be at their highest and expected to have health implications. Concentrations can be predicted for the future, taking into account emission controls and new or changed source emissions (Stedman et al., 2007). The dispersion process is however subject to inherent uncertainties, in part due to the complexity of the various sources involved and the difficulty in measuring such factors. As a result, there are a number of different air quality models for predicting pollution concentrations in urban environments, with varying rates of accuracy in different scenarios (Gokhale and Raokhande, 2008; Patra et al., 2008). There would appear to be a need for a more systematic placement of monitors in a dense grid on a national level, to ensure sufficient data sources. These would yield more accurate data mapping and provide quantitative data to accurately appraise current methods used in modelling $PM_{2.5}$ concentrations for air quality assessment in the UK (Stedman

et al., 2007). Mathematical dispersion models can be used to assess air quality in areas where monitoring data is not available, or where fine detailed assessment is required at a specific location.
Boroughs and Council administrations often use the Gaussian-type plume model (ADMS) for local air quality forecasting, however questions have been raised over its accuracy and ability to measure boundary layers in an urban situation. Using pulsed Doppler LIDAR data to measure mixing layer heights and cloud base, Davies et al., (2007) discovered inaccuracies of 30-100% in the predictions of the Met Office unified model (UM), also that the ADMS model was poor in predicting ABL heights in any urban situation.

Generalised additive modelling, which looks at comparing both linear and non-linear variables, has shown that the most important factors in predicting overall $PM_{2.5}$ concentrations, are those related to traffic volume and wind. Low winter temperatures, particularly those below 0° C also play a part in increasing concentrations of $PM_{2.5}$ in particular. Other factors include relative humidity and precipitation (Aldrin and Hobeak Haff, 2005). A readily available computer programme that incorporates these parameters is the Operational Street Pollution Model (OSPM), a street pollution model developed by the National Environmental Research Institute (NERI) and extensively tested (Kukkonen, et al., 2003; Vardoulakis et al., 2007). OSPM includes a Gaussian plume model for direction contribution of $PM_{2.5}$ and a box model for the street canyon and recirculation of pollutants in the street. The influence of traffic induced turbulence on pollutant dispersion is also taken into account. For both the current and future scenarios, hourly input files of wind speed direction, temperature and global radiation plus background $PM_{2.5}$ concentrations are required. All inputs required are shown in Figure 3.4.

OSPM provides an operational pollutant modelling software, with all the data inputs readily available to enable quantification of $PM_{2.5}$ concentrations within streets in the GLA for 2010 and 2050 scenarios. This can be used to predict variations of concentration within the GLA based on distance from a main road and or with height within a street, helping to confirm the impacts of location relative to pollutant source within an area. The various inputs needed to operate OSPM (as used in this study to analyse relative locational impacts, see section 5.1.6) and are readily available, are seen in Figure 3.4

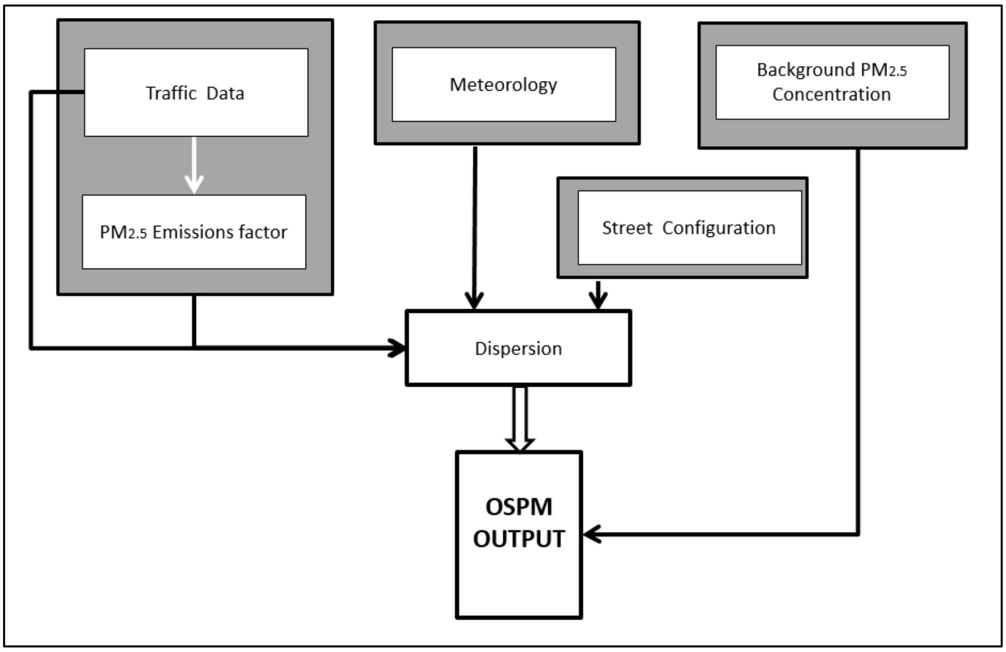

Figure 3.4 Modelling inputs for OSPM

In the future, external concentrations of $PM_{2.5}$ are expected to decline due to policy impacts leading to reductions in vehicle exhaust emissions and gaseous precursors which produce secondary particles (Williams, 2007). However, this is by no means certain and as such future modelling needs also run scenarios with a business as usual concentration of external $PM_{2.5}$, where policies are ineffective in kerbing emissions.

3.5 $PM_{2.5}$: Indoor Sources and Sinks

Indoor $PM_{2.5}$ concentrations have been linked to transient emissions from internal sources such as construction materials, fixtures, fittings and appliances as well as intermittent emissions such as the burning of fuels and candles, smoking, cooking, heating and human domestic activities (Milner et al., 2005; Weschler, 2009). Studies have shown high $PM_{2.5}$ indoor concentrations relative to external levels, with cooking and smoking being the two primary sources (Jones et al., 2000). Occupant movement and behaviour, including window opening, can affect indoor concentrations (Andersen et al., 2009). Within dwellings, modelling suggests that different rooms could be subject to very different levels of $PM_{2.5}$, depending on the activities conducted in them (Dimitroulopoulou et al., 2006). Methods are needed to measure and assess the impact of cooking, smoking and domestic activities as well as ventilation behaviour in order to fully understand both the current stock level concentrations and the impact of energy efficient refurbishment on future concentrations. Numerous studies, including some based in the UK, have shown relatively high $PM_{2.5}$ indoor emission rates from which emissions inventories can be composed (Jones et al., 2000; Choa and Wong, 2002; Sawant et al., 2004). Whilst this will give the concentration within a room or group of rooms in a house, the actual exposure experienced by an occupant is calculated by their movement within a property (and elsewhere) and the time they are exposed to various concentrations of $PM_{2.5}$. Additional to emissions from indoor sources, concentrations of $PM_{2.5}$ in houses are affected by the infiltration of outdoor particles and the removal from the internal air by deposition, filtration and exfiltration, though some re-suspension also occurs largely related to domestic activities (Gehin et al., 2008). In apartments (as well as terraced and semi-detached houses), inter-dwelling transfer of contaminants via party wall permeability is also possible (Molnár et al., 2007). Increased air tightness and installation of mechanical ventilation and heat recovery systems (MVHR) which filter out $PM_{2.5}$, could reduce the penetration of externally generated $PM_{2.5}$ into dwellings. However, any increase in air-tightness without an increase in controlled purpose provided ventilation could lead to a rise in exposure from internally generated $PM_{2.5}$ (Wilkinson et al., 2009). $PM_{2.5}$ has been linked to internal sources such as materials used in building construction, fixtures and fittings, appliances, cooking, as well as a variety of domestic activities smoking, (Meng, et al., 2009; Weschler, 2009). All these main sources should be included in any modelling scenarios, although smoking and non-smoking scenarios should be run separately so as to distinguish smoking households from non-smoking households in order to estimate the likely general background concentrations of $PM_{2.5}$ in the domestic environment. Emission rates for cooking derived from the PTEAM study, the largest study of its kind (Ozkaynak et al., 1996) which showed a rate of 4.1+ 1.6 mg min-1 of inhalable PM10 of which 40% is the finer fraction of $PM_{2.5}$ (1.6+ 0.6 mg min^{-1}). Deposition of $PM_{2.5}$ is based on published data of deposition velocity which is dependent on the surface to volume ratios of the individual rooms (Ozkaynak et al., 1996). This study gave mean deposition velocities of 1.8 x 10-4 ms-1, although it is acknowledged that in reality, room furnishings, air speed and particle spatial distribution can affect particle deposition rates and there is therefore a need for sensitivity analysis to be carried out on deposition rates (Thatcher et al., 2002; Zhao and Wu, 2009). In addition, there is intense debate as to the differences in the nature of the particles of indoor and outdoor origin and their potential health impacts. Although largely unqualified, the current debate suggests that for now, the two sources should be treated separately in any modelling (Adgate et al., 2007).

3.5.1 Measuring Indoor $PM_{2.5}$

There are numerous studies that measure indoor $PM_{2.5}$, using a variety of equipment and differing methodologies. Equipment used for indoor domestic and external monitoring of $PM_{2.5}$ emission sources and concentrations are reviewed and assessed on the basis of their performance in experiments. Monitors for indoor domestic use are usually small or handheld for short-term monitoring where space is at a premium. Measurements are taken either of concentrations in µg m^{-3} or emission rates in either µg min^{-1} or mg min^{-1} (He et al., 2004). External monitoring, is often mandatory as a result of a statutory duty and requires large and complex equipment that can be left to monitor continuously for long periods and can provide hourly, daily, and annual concentrations, measured in µg m^{-3} (UKAQA, 2010).

3.5.2 Equipment for Indoor Measurement

ISO 16000-1:2004 is the main European standard intended to aid the planning of indoor pollution monitoring, offering suggestions on the development of a suitable sampling strategy. However, a strategy specifically for $PM_{2.5}$ is not included and may explain the diverse approaches to this issue in the literature (ISO, 2010). Positioning of monitors relative to the source varies between studies, although in the majority of literature this is not listed and it is unclear as to how this may affect the readings. He et al., 2004 placed the instruments on average 2m from the stove in the kitchen, Chen et al., 2009 4m away. Choa et al., 2002, placed the meters 1.1 m above the ground to simulate the breathing zone of the average human (although this appears a little low in my opinion) and went on to infer stratification in pollutant concentration in domestic properties, where low turbulence occurred and therefore poor mixing. See and Balasubramanian, 2008 positioned theirs ~0.2m from the cooking source and at a height of ~1.5 metres, as this was thought to more accurately represented the breathing zone of the cook. Indoor combustion sources, specifically gas cookers, potentially represent a high level $PM_{2.5}$ source (Ozkaynak et al., 1996). Protocols for monitoring CO sources from gas cookers using portable analysers is covered by BS EN 50291:2001. It would seem appropriate that these protocols could provide a basis to positioning monitors for $PM_{2.5}$ as well.

The use of gravimetric impactors and light-scattering equipment are the two methods most widely used to measure indoor $PM_{2.5}$ concentrations, although impactors can be noisy and bulky (Chen et al., 2009). These instruments are fundamentally different in their function and as such have been the subject of comparative studies, which have emphasized the need for correct calibration to avoid erroneous results. Findings using both methods are generally similar where well-defined and strictly controlled conditions occur, although light -scattering devices tend to give readings above that of gravimetric equipment (Tasic et al., 2012). This led Niu et al., 2002 to conclude that light-scattering devices should only be used as preliminary screening instruments with greater credibility given to impactors as one of the most sensitive instruments available. This is especially so for changes over time and the chemical characterisation of indoor/outdoor $PM_{2.5}$ relationships, although light scattering devices can fulfil this function if additional filtering capacities are used and quantify elemental composition (Chao et al., 2002). Systems of measurement vary with both concentration and emission rates being measured depending on the focus of the study (Huang et al., 2004; Tang et al., 2007).

Gravimetric impactors use pumps and filters to collect samples of specific sizes, which are analysed gravimetrically using a micro-balance with concentrations calculated using mass-balance equations (Moschandereas et al., 1987). The device can be configured to collect particles of a specific size, or by removal of the impactor, total suspended particulates. Air is drawn into the sample inlet at a constant specified flowrate where it passes through the particle size separator (impactor) and on to the collection filter. This sample can then be weighed and compared to the pre-weighed filter to obtain PM mass (Niu et al., 2002).

In respect of this current study, one of the issues with impactors is that they do not give a real-time estimation of pollutant emissions. If they are run in an attempt to obtain estimated mass from a short-term emission source, the low mass obtained has been found to produce significant weighing errors, which are also evident with low concentrations of particulate matter (Niu et al., 2002). Loss of the more volatile species also occurs, especially where the device is run for extended periods and gravimetric equivalent calculations must be used to compensate for this (Sloss and Smith, 2000). Filters used in the equipment are usually preconditioned and dried; this can result in humidity differences compared to the monitored environment which can lead to condensation and possible erroneous readings if this is not compensated for. Gravimetric impactors are not considered ideal for measuring individual emission sources, although they can give accurate results for the distribution of $PM_{2.5}$ components (Liu et al., 2002). These factors may help to explain the slightly lower readings recorded when compared to Light-scattering devices

Light-scattering devices, such as the DustTrak range, calculate real-time volume concentration of aerosol particles. An air sample is continuously drawn via the inlet into an internal chamber where a laser source illuminates the stream of particulate matter. The light is scattered in all directions and some is collected and focused onto a photo detector where it is converted into a voltage signature, which is proportional to the amount of scattered light measured and yields the mass concentration of particulate matter. He et al., 2004 found that there was a time delay in the increase of PM2.5 compared to the increase in concentration of sub-micrometre particles, likely to be caused by the coagulation of aerosols and the shift of particle size distribution with time. The concern here is that although particle mass in reality remains constant; as the DustTrak detects larger particles with greater efficiency, the measured increase in $PM_{2.5}$ may not be consistent with the figures for total particle mass. The advantage of machines such as the GRIMM 1.108 is that it incorporates a collection filter, such that mass balance confirmation can occur within the machine (Grimm, 2010). However, this is subject to a protected algorithm and such may prove difficult to confirm externally. Equipment manufactured in the USA tend to use Arizona Road Dust (A1) as its calibration default, but if gravimetric sampling within the device is used a custom reference calibration, correct to the area being studied can be achieved with a flow accuracy of +/- 5% of the factory setpoint (TSI, 2010)

Other light-scattering equipment used includes a Nephelometer, which measures suspended particle density in any gas or liquid. The results obtained are defined by particulate size, colour and shape. However, empirical evidence suggests a differential response, with particles 0.3-2 µm being more effectively detected relative to the size fraction 2-10 µm (Breysee et al., 2005). This could lead to incorrect estimates of $PM_{2.5}$ concentrations and mass distribution in the ambient air if not recognised and compensated for.

He et al., 2004 used a condensation particle counter (CPC), which measures the total number concentration of particles by growing the ultra-fine particles through a condensing process in a vapour carrier. From a total range the sub micrometre sized particle number must be approximated. In this instance a DustTrax monitor was run alongside this to enable confirmation of the $PM_{2.5}$ particle mass. The results obtained, suggest a close correlation between the two devices; however, Nephelometer and CPC equipment are bulky and probably unsuitable for most indoor domestic environments.

Yanosky et al., 2002 conducted a comparison of an Aerodynamic Particle Sizer (APS) against a DustTrax monitor and compared both sets of results to the US EPA designated Federal Reference Method (FRM). The APS accelerates the particulate sample flow through an orifice and determines the aerodynamic size of a particle by its rate of acceleration. It measures in real-time enabling collection of information on particulate number, mass concentration and size distribution on small time scales. Results showed the DustTrax levels were well correlated with the FRM ($R2= 0.859$), but with a significant proportional bias ($\beta1=2.57, p<0.001$). Accuracy was improved by statistical adjustment. The APS showed similar results, however TSI, the manufacturers acknowledged errors in the equipment after it was discovered that small particles were recycling at slower velocities causing additional phantom readings of larger particles. These errors have been corrected in more recent models such that there is greater correlation between studies and the proportional bias has been removed (TSI, 2008).

The use of more than one device in measurement may be initially necessary to confirm accuracy, with gravimetric impactors and light-scattering equipment - the two methods most widely used. The most important issue is correct calibration, with the factory pre-sets not necessarily being appropriate to all situations. Consultation with manufacturers may be necessary prior to measurements commencing (Jantunen et al., 2002). In deciding which monitors to use, room layout, positioning of the equipment and risk assessments are important inputs into the decision process. In mE modelling, a uniform dispersal of pollutants within a zone is automatically assumed as in the case of pollutant models such as CONTAM and EnergyPlus used in this study. This can add to issues of uncertainty in the modelling that need to be addressed by carrying out sensitivity analysis (Milner et al., 2011). Monn et al., (1997) debate the effectiveness of single monitors in estimating the personal exposure to $PM_{2.5}$, pointing to modelling validated by quantitative data being the key to successful exposure profiling. In small domestic environments it would appear that hand held or surface mounted light scattering devices are the most practical for measuring $PM_{2.5}$ (TSI, 2010).

3.5.3 Legislation and Policies Affecting Indoor Domestic $PM_{2.5}$

There is currently no direct legislation covering $PM_{2.5}$ in domestic indoor environments in the UK, which would be both hard to frame, monitor and enforce. It has been previously noted that even where such legislation exists (in terms of external ambient concentrations), such legislation is hard to enforce. Ambient limit values recommended by the World Health Organisation are an annual mean of $10\mu gm^{-3}$ with a 24-hour mean of $25\mu gm^{-3}$, although current epidemiological evidence suggests there is no safe limit (WHO, 2006). ISO 16000-1:2004, the European standard intended to aid the planning of indoor pollution monitoring does not include a strategy specifically for $PM_{2.5}$ and may explain the diverse approaches to this issue in research (ISO, 2001).

In the context of climate change, the twin issues of the need to reduce airborne pollution (including greenhouse gas emissions (GHG)) and the protection of human health have become intertwined. The UN Convention on Climate Change (2009) which the UK has signed and ratified is the primary international mechanism to establish legally binding commitments to reduce GHGs. The European Council in March 2007 approved an ambitious energy package on GHG emissions, bio fuels and renewable energy that represents the decision to shift from the current, primarily carbon based energy economy (which is reliant on fossil fuels), to cleaner, primarily renewable energies and energy efficiency measures (a low carbon economy) (LCE) (CEU, 2007; Hopkins, 2008; DEFRA, 2009). The UK Government has accepted the recommendations of its Committee on Climate Change (CCC) for a reduction of GHG by 80% by 2050 relative to 1990 levels, taking a lead role in trying to produce a healthier environment and securing a reduction of airborne pollutants which will include $PM_{2.5}$ (CCC, 2014). Additional policies, strategies and mechanisms (See section 2, table 2.1) were enacted which would influence IAQ, although as previously stated some measures (e.g. the Green Deal) have since been cancelled. Both new-build and the retro-fitting of existing properties will be affected by changes in the Building Regulations Parts L and F. This will set standards for CO_2 emissions, as increasing insulation and air tightness and may affect internal domestic $PM_{2.5}$ levels. Additionally, the use of purpose provided ventilation (PPV) and mechanical ventilation systems may increase or decrease the ingress of external pollutants and reduction of indoor concentrations.

Even though, in developed countries people are increasingly spending more time in indoor environments 85-90 % (Klepeis et al., 2001), there is currently no legislation covering $PM_{2.5}$ in domestic indoor environments. The UK ISO 16000-1:2004 covers domestic indoor environments, but only deals with a strategic approach to sampling and does not include $PM_{2.5}$ in its remit (ISO, 2010). Much European legislation for indoor air quality exists e.g. ISO TC 146 (air quality) SC6 (indoor air) and for control of hazardous substances, many of which may be found in domestic dwellings. However, these regulations only apply to the workplace. The ambient limit values recommended by the World Health Organisation are an annual mean of $10\mu gm^{-3}$ with a 24-hour mean of $25\mu gm^{-3}$, although current epidemiological evidence suggests there is no safe limit (WHO, 2006; DEFRA, 2010).

Numerous studies (including some based in the UK), have shown relatively high $PM_{2.5}$ indoor emission rates, from which emissions inventories can be composed (Jones et al., 2000; Choa and Wong, 2002; Sawant et al., 2004). This problem is acknowledged with the Committee on the Medical Effects of Air Pollutants (COMEAP) and the WHO providing guidelines on safe levels of some indoor pollutants, as well as providing guidance on management of indoor air quality (COMEAP, 2004; WHO,2006). However, COMEAP have stated that were a standard to be in place for $PM_{2.5}$, monitoring every home would be impractical and restricting people's freedom to act as they wish may not be possible (COMEAP, 2004). Unlike some other indoor pollutants, it is believed not to be currently feasible to define a satisfactory guideline for indoor particulates. This is based in part on the lack of sufficient health evidence (specifically for indoor $PM_{2.5}$) and in part due to the lack of clear characterisation of indoor particulate matter, which is perceived to have different sources, chemical compositions and size distribution from outdoor aerosols. Some locations where properties have high air change rates may have similar profiles and concentrations to outdoor particulate matter. It is also acknowledged that indoor concentrations may in some cases exceed those found outdoors. The WHO state that they believe the health effects of the same pollutants indoors will be similar to those from outdoor studies, with the guideline development sub-committee stating that *"the current air quality guidelines for PM provide targets which are also valid for indoor environments"* (COMEAP, 2004; WHO, 2006). As no actual standards for indoor $PM_{2.5}$ are suggested, the COMEAP report focuses on 3 spheres of influence to achieve changes in pollutant concentrations and indoor sources. Material or device managers, architects and building engineers (who had influence over building ventilation design) and individuals with concerns, who could carry out monitoring and need a bench-mark to compare obtained results against.

The COMEAP guidance to minimise general pollutant concentration covers four factors:

- The outdoor concentration of air pollutants;
- The extent of filtering (attenuation) imposed by the building on air passing from outdoors to indoors, indoor adsorption/desorption and chemical reactions;
- The indoor sources of the pollutants;
- The level of ventilation of the building.

These represent the primary sources, sinks and fluxes. The WHO identified a number of risk assessments and reviews of indoor pollutants, including $PM_{2.5}$, which had sufficient evidence to enable the development of indoor air quality (IAQ) guidelines. However, the latest WHO publication does not include $PM_{2.5}$ as a source to be evaluated (WHO, 2010). It would appear that, within the development of the guidelines, $PM_{2.5}$ has been characterised in terms of a source from indoor combustion only (Krzyanowski et al., 2006).

The overall legislative approach to external and indoor $PM_{2.5}$ is therefore very different, as legislation for acknowledged indoor levels may encroach on personal liberty and be hard to frame and enforce. This is in part, due to a lack of epidemiological evidence linked exclusively to indoor $PM_{2.5}$ and the ongoing debate previously mentioned regarding the suggested differences in relative toxicity of $PM_{2.5}$ from outdoor and indoor sources. Specific influence has been exerted on material manufacturers, to reduce products that are likely to create air quality issues. Building designers are encouraged to ensure adequate ventilation, although behavioural factors may counteract this (COMEAP, 2004; WHO, 2006).

3.6 $PM_{2.5}$: Outdoor/Indoor Exchange

In order to distinguish particles of external and internal origin, enrichment factor (FE) analysis can be carried out. This form of analysis assists source appointment study, by providing an initial assessment of whether indoor particulate matter comes from the Earth's crust, industrial process or indoor activities. Comparisons of elemental concentrations of $PM_{2.5}$ against natural background levels will yield factors of approximately 1 if there are no additional anthropogenic contributions to $PM_{2.5}$ levels. High EF indicates the species probably comes from industrial sources, vehicle exhaust, indoor combustion related or other human activities. Choa and Wong, (2002) noted that when comparing $PM_{2.5}$ and PM_{10} from indoor domestic environments in Hong Kong, the fine mode species were highly enriched with bromine, lead, nickel, potassium, sulphur, vanadium and zinc, while species in coarse mode had an enrichment factor close to 1. This seems to infer that $PM_{2.5}$ species contain elements that are more likely of manmade origin. However, Ravindra et al. (2008), whilst acknowledging this to be the case, points to the greater ability of smaller particulates to penetrate the building envelope from outside sources. A clear distinction between outdoor and indoor particulate generation and elemental basis is required to confirm source apportionment and factors influencing indoor concentrations, including Indoor/ Outdoor (I/O) ratios (Gehin et al., 2008) and also in light of the ongoing debate regarding the (assumed) relative differences in toxicity of $PM_{2.5}$ from indoor and outdoor sources. Air change rates, often measured by the use of tracer gas decay rates are used to confirm analysis and provide a statistical correlation (Chen et al., 2009). There is some debate over the use of carbon dioxide (CO_2), as it has many internal and external sources that need to be accounted for, with some studies using other gases such as sulphur hexafluoride (SF6), although this is no longer used (Sawant et al., 2004; Choa and Wong, 2002) to calculate air change rates.

Air change is one factor influencing Outdoor/Indoor (I/O) ratios of $PM_{2.5}$. As previously shown the different fractions of $PM_{2.5}$ vary in their ability to penetrate the building envelope (penetration factor-p), with attenuation of the larger factions being more likely through smaller crack and advantageous openings. However, for purpose provided ventilation (PPV) that is unfiltered, e.g. open windows, the factor is near to 1. In buildings with air conditioning or during the heating season, windows are often closed. Compared with natural ventilation, this reduction in ventilation results in a relatively low air exchange rate. Yamamoto et al. (2010) examining 500 domestic properties in the US reported a mean air change rate of 0.71 ACH (air changes per hour), and a lower I/O ratio where infiltration was the primary pathway for $PM_{2.5}$ entering dwellings. Chen and Zhao (2011) carried out an analysis of over thirty studies from around the world, each where more than twenty properties were studied. In addition, they considered previously published studies to both measure and model the infiltration/penetration factor. They concluded that the penetration factor (p) (the ability of $PM_{2.5}$ to penetrate the building envelope) had a range of (1.0-0.6). As these values greatly impact on indoor concentrations and therefore exposure to indoor $PM_{2.5}$ from outdoor sources in modelling, it seems prudent that they should be included in any sensitivity analysis.

3.7 Deposition of $PM_{2.5}$

Deposition is the process by which aerosol particles deposit themselves onto solid surfaces, and in conjunction with exfiltration is the primary mechanism for decreasing the concentration of particles in the indoor air. The rate of deposition or *deposition rate (Dr)* has been defined as the number of particles deposited per unit surface area with time in $m^{-2}s^{-1}$ (Thatcher and Layton, 1995). The *deposition velocity (Dv)* is defined as the *deposition rate (Dr)* divided by the undisturbed concentration and is slowest for particles of an intermediate size, with mechanisms for deposition being most effective for either very small particles (<1µm) or very large particles (>10 µm) (Chen and Zhoa, 2011). Very large particles will settle out quickly through gravitational sedimentation (settling) or impaction processes, while Brownian diffusion has the greatest influence on small particles (Seinfeld and Pandis, 2006). Another term *deposition loss rate coefficient/ deposition loss rate or deposition rate constant* is less frequently used and refers to the number of particles depositing on the total available surface with time in s^{-1} (Morawska and Salthammer, 2003).

The *deposition rate (Dr)* for $PM_{2.5}$ is calculated using the following equation

$Dr = Dv\ (A/V)$ Equation 1

Where:

Dr= Deposition rate in $^{1}/s$
Dv= Deposition velocity in m/s
A= Surface area of a given space in m^2
V= Volume of a given space in m^3

However, the literature gives a variety of preferences as to what 'A' values should be used, with some authors linking this to particular $PM_{2.5}$ fractions and surfaces within a room and others using the whole surface area of a room (e.g. Thatcher and Layton, 1995; Fogh et al., 1997; Thornburg et al., 2001). Resuspension can also occur, primarily through various human activities such as domestic cleaning and walking (Gehin et al., 2008).

3.8 Summary and Implications for Research

This chapter considers the many factors influencing indoor $PM_{2.5}$ concentrations. It establishes the characteristics of $PM_{2.5}$ and explains the resulting health impacts. It establishes the main outdoor and indoor sources and processes of removal, their methods of measurement and issues arising from measurement methods, factors affecting their concentrations and details the current legislation and policy tools applied to particulate matter. It then outlines the modelling methods currently used to establish the urban spatial distribution of $PM_{2.5}$ concentrations, based on monitored large scale data of external concentrations over time. Following this investigation, the processes whereby outdoor $PM_{2.5}$ penetrate the building envelope are discussed, as well as the methods of removal from the indoor air. Current research shows that there are a wide range of values for emission rates, deposition rates and building penetration factors (p) of $PM_{2.5}$ material and a level of uncertainty exists. For this reason, a thorough sensitivity analysis needs to be carried out in modelling to show the distribution of possible indoor $PM_{2.5}$ concentrations. The material from the chapter helps to influence the methodology seen in the following chapter. A number of issues reviewed in this Chapter have implications on the methodological approaches utilised in this thesis: this will be discussed further in the next Chapter(s).

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Chapter 4

Overview of Methodology

Introduction

The literature review so far has highlighted potential changes in future indoor domestic exposure to $PM_{2.5}$ and its associated impacts on population health as a prominent unintended consequence of decarbonisation of the domestic stock, and described how the scale of such impacts is currently unclear. This chapter outlines the underlying physics impacting the movement and concentrations of $PM_{2.5}$ in dwellings, and a broad overview of the modelling methods deployed in this study. It reviews relevant modelling approaches, discussing the reasons for selecting those utilised in this study. An outline of the basis of the assumptions used in the development of the models is also provided. The function of this chapter is to give a clear understanding of the modelling techniques and inputs used in this study, which will aid understanding of the research that follows. Further methodological details specific to individual investigations are given within the relevant Chapters (5-7); these subsequent chapters also detail any variations in methodologies and modelling approaches according to the particular focus of the research question being asked. This includes the expansion of the building stock model to be representative of both regional and national stocks and the use of coupled dynamic temperature-pollutant models.

A number of empirical studies have been conducted worldwide using different measurement methods, in order to quantify indoor $PM_{2.5}$ concentrations and derive associations with occupant, environmental, or building characteristics. Studies have used either personal monitoring equipment or static systems within buildings to examine the general association between indoor $PM_{2.5}$ concentrations and personal exposure. Hanninen et al. (2004) as part of the EXPOLIS project monitored average indoor $PM_{2.5}$ concentrations in non-smoking households in four European cities, showing the following variations: Athens 23+11μg.m^{-3}; Basle17+8μg.m^{-3}; Prague 25 +16μg.m^{-3} and Helsinki 25+16μg.m^{-3}. Wallace et al. (2006) monitoring 36 residences in North Carolina over a year show mean indoor $PM_{2.5}$ concentrations of 25.8 μg.m^{-3} with a range of 7.2-66.0 μg.m^{-3} for non-smoking households. Substantial variations are seen in empirical studies on smoking concentrations including $PM_{2.5}$. For smoking properties, a consumption of 7.4 cigarettes per day results in an average indoor $PM_{2.5}$ concentration of 132.7 μg.m^{-3} measured over 14 days while 4 cigarettes over 19 days yields 66.0 μg.m^{-3} (Wallace et al., 2006).UK specific studies have compared indoor $PM_{2.5}$ levels in dwellings in roadside, urban, and rural situations (Jones et al., 2000) and also showed cooking and smoking were determined to be the major indoor sources of $PM_{2.5}$, whilst cleaning and general activity had little influence on concentrations. Dimitroulopoulou et al. (2005) in a monitoring study on kitchens in 37 new homes in the UK with smokers found 24-hour mean $PM_{2.5}$ concentrations of 113μg.m^{-3} in winter and 134μg.m^{-3} in summer Wheeler et al. (2000) quantified seasonal variations in indoor $PM_{2.5}$ concentrations during winter, spring and summer which were 22, 17 and 18 μg.m^{-3} respectively. Mohammadyan, (2005) showed average indoor domestic concentrations of $PM_{2.5}$ of 19.0 μg.m^{-3}. In North East Scotland, Osman et al. (2007) reported that average indoor $PM_{2.5}$ levels were 18 μg.m^{-3}. In Manchester, a study by Gee et al. (2002) showed that the levels of indoor $PM_{2.5}$ (5-day mean) in living rooms and bedrooms were 28.4 μg.m^{-3} and 19 μg.m^{-3}, respectively. Nasir and Colbeck (2013) examined $PM_{2.5}$ concentrations inside different dwelling types, although the role of building type was not clear due to variations in occupant behaviour.

Empirical studies like these require considerable resources and a large number of participants or locations in order to be statistically significant. Consequently, such studies are generally small and are often only applicable to the geographical location studied, or the individual aspect of $PM_{2.5}$ research being investigated.

Alternatively, if sufficient input data exists, a scenario-based exposure modelling approach is a cost effective and efficient tool for quantifying the many factors that affect personal exposure to $PM_{2.5}$ and also exploring the future scenarios addressed in this study (Johnson, 2001). Multizonal modelling has been previously used as a means of investigation of indoor $PM_{2.5}$ concentrations. Dimitroulopoulou et al. (2006) used the INDAIR probabilistic model to calculate annual indoor mean $PM_{2.5}$ concentrations in households cooking with gas. Fabian et al. (2012) used CONTAM to model indoor $PM_{2.5}$ concentrations from cooking in low-income multifamily housing. Emmerich and Howard-Reed, (2005) using CONTAM modelling as part of the U.S Dept. of Housing and Urban Development's Healthy Homes Initiative, found the two most effective intervention strategies for indoor air quality were extract fans and (HVAC) systems.

4.1 Fundamentals of Airflow and Pollutant Transport

This section outlines the underlying building physics impacting the movement and concentrations of $PM_{2.5}$ in dwellings that need to be accounted for in any modelling software used to calculate changes in indoor $PM_{2.5}$ concentrations.

4.1.1 Building Envelope Permeability/Airtightness

Building airtightness (also called envelope airtightness) is defined as the resistance to inward or outward air leakage through unintentional leakage points/cracks in the building envelope, as opposed to purpose provided ventilation (PPV) such as windows or ventilation systems. The air leakage is driven by differential pressures across the building envelope (external faces), which can be caused by wind driven and/or buoyancy effects. The relationship between leakage air flow rate and pressure is subject to the 'power law', between the airflow rate and the pressure difference across the building envelope as shown in equation 2:

$$Q_L = C_L \Delta p^n \qquad \text{Equation 2}$$

Where:

Q_L= the volumetric leakage airflow rate expressed in m^3/hr

C_L= the air leakage coefficient expressed in $m^3/hr/Pa^{-n}$

Δp= the pressure difference across the building envelope expressed in Pa (usually 50Pa).

n = the airflow exponent ($0.5 \leq n \leq 1$) 0.5 for fully turbulent flow and 1 for purely laminar flow.

The airtightness of a building may then be expressed using a number of metrics. Once the leakage airflow rate through the building's envelope is known, at a given reference pressure (usually 50Pa) it can be either; divided by the heated building volume to give the ‘air change rate’ (hr); by the floor area to give the ‘specific leakage rate’ ($m^3/hr/m^2$), or divided by the envelope to yield the ‘air permeability’ ($m^3/hr/m^2$). The latter is the metric most commonly used in UK regulations, and in this study.

The current Building Regulations Approved Document Part L1A (2010) stipulates testing for new builds, carried out by using a ‘blower door test’ in accordance with BS EN 13829 (2001). Part L1A requires an air tightness of maximum 10 $m^3/hr/m^2$ air loss at a pressure of 50 Pa. Standard good practice for air tightness testing in the UK is a maximum of $7m^3/hr/m^2$@50Pa and best practice is $3m^3/hr/m^2$@50Pa (ADL,2010). This standard measures air permeability in $m^3/hr/m^2$@50Pa, the air leakage per m^2 of building envelope (AD, 2010), with current permeabilities in the UK stock ranging between 3-30 $m^3/hr/m^2$@50Pa (Stephen, 1998, 2000). In addition, purpose provided ventilation (PPV) such as windows, trickle vents and mechanical ventilation e.g. extract fans, HAC and MVHR systems contribute to the hourly air change rate.

4.1.2. Infiltration Rate and Air Changes

The infiltration rate (IR) is the volumetric flow rate of outside air into a building, measured in litres per second (l/s). The air exchange rate (ACH), is the number of interior volume air changes that occur per hour, including all forms of infiltration via the building envelope (cracks and gaps), as well as window and door openings and any passive or active ventilation supplied to the building.. Building Regulations, specifically ADF, 2010 Table 5:1b stipulates the whole dwelling minimum ventilation rates (l/s), which increase dependant on the number of bedrooms in the property and also increases when more than two occupants are living at the property (AD, 2010). ACH can be calculated by dividing the building's IR by the building volume.

$$ACH = [(IR/1000)3600]/V \qquad \text{Equation 3}$$

Where:

ACH= Building air change rate in m^3/hr
IR= Infiltration rate in l/s converted to m^3/hr
V= Internal building volume in m^3

4.1.3 Drivers of Airflow

Buoyancy driven ventilation can arise due to differences in density of indoor and outdoor air which in large part arises due to differences in temperature and moisture content. When there is a temperature difference between two adjoining volumes of air the warmer air will have lower density and be more buoyant thus will rise above the cold air creating an upward air stream. In buildings, the greater the thermal and moisture differences and the height of the structure, the greater this ‘stack effect’, which helps drive natural ventilation and infiltration of air and the movement of airborne pollutants such as $PM_{2.5}$ both into and out of the dwelling. The stack effect or draft-flow rate is given by:

$$Q = CA\sqrt{(2gh(\frac{Ti-To}{Ti}))} \qquad \text{Equation 4}$$

Where:

Q= Stack effect (draft-flow) rate in m^3/s
A= Flow area, m^2
C= Discharge coefficient (0.66)*
g= Gravitational acceleration, 9.81 m/s^2
h= Height or distance, m
Ti= Average inside temperature, °K
To= Outside air temperature, °K
* assumed to be 0.66 for gaps and cracks (Fang and Persily, 1995).

Wind patterns around buildings in built-up areas can be quite complex. Where wind strikes a building perpendicularly it exerts a positive pressure on the facade, that can lead to negative pressures on the leeward side. This pressure decreases as the angle of wind flow moves from the perpendicular for a given wind speed. This wind creates a pressure difference across the building façade and drives airflow into the building both though unintentional leakage points/cracks in the building envelope and any purpose provided ventilations. These forces drive the movement of airborne pollutants into and out of buildings.

4.1.4. Pollutant Infiltration and Deposition of $PM_{2.5}$

Issues of infiltration and deposition were dealt with under sections 3.6 and 3.7 above. However, whilst it is acknowledged that the various particle sizes within $PM_{2.5}$ behave differently in terms of deposition processes, for the purpose of this study, $PM_{2.5}$ has been considered as a whole in line with similar previous monitoring studies (Dimitroulopoulou et al., 2006; Wilkinson et al., 2009). This decision is also influenced by modelling restraints, which only allow for one deposition rate to be used for all surfaces, and also due to the different approaches/opinions expressed by various authors as previously discussed.

4.2 Methodological Approach: Broad Overview

Following the literature review and the collection of various input data for the modelling; the overall methodological approach could be summarised as follows:

i) The selection of construction geometries (archetypes) to represent the London, Milton Keynes and whole English domestic stock (i.e. building stock modelling, built form data; such as geometry, archetype, ventilation characteristics etc.) and selection of data inputs.

ii) Construction of archetypes in multi-zone airflow and pollutant transport simulation models (CONTAM & EnergyPlus). Creation of a range of variants of the archetypes, with different permeabilities and addition of various ventilation and energy efficiency components.

iii) Investigation of the stock profile of the location/s being researched in order to scale-up the modelling to be representative of the built stock.

iv) Explore current and possible future scenarios in order to establish annual average indoor $PM_{2.5}$ concentrations, the distribution of values and evaluate modelling uncertainties through sensitivity analysis.

v) Further analysis of results in order to quantify the possible impacts of occupant behaviour, income and location, following the installation of various ventilation and energy efficiency strategies resulting from Government policies to de-carbonise the built environment. Figure 4.1 shows the outline methodology used.

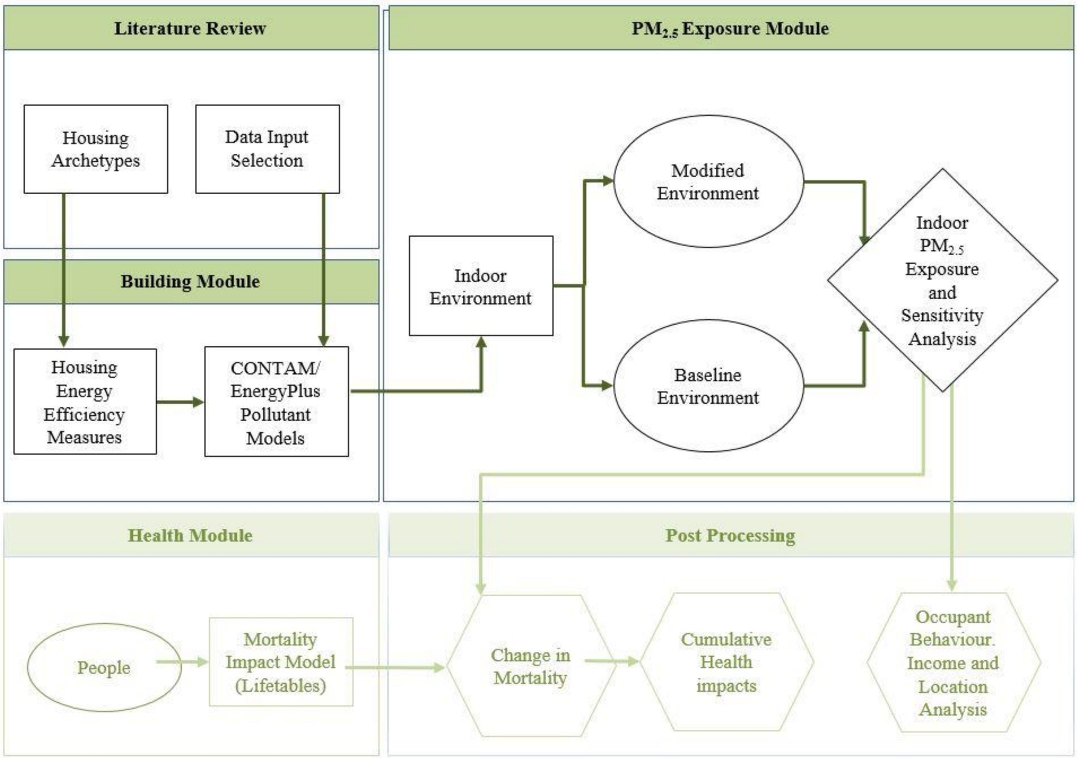

Figure 4.1 Outline methodology. Darker parts of the graph are indicative of general methodology (Chapter 4), whilst shaded components show investigations seen in Chapters 5-7.

4.3 Methods for Modelling Indoor $PM_{2.5}$ Exposure

There are a variety of indoor exposure modelling techniques and software available to investigate the impacts of changes in $PM_{2.5}$ at different scales and levels of accuracy. These range from simple statistical regression and mass balance approaches, to more complex multizone and computational fluid dynamics tools that have large and complex input data requirements which demand greater computer processing time. A brief description of each model types follows, with the basis of selection of programmes for this study.

4.3.1 Statistical Regression:

For this method, linear and nonlinear regression techniques are used to relate indoor exposure to its determinants based on measurement data. Regression methods require indoor personal exposure data and corresponding observations, such as outdoor pollution concentrations, indoor source strengths and housing characteristics. The resulting equation obtained by any of the regression methods may then be applied to model indoor exposure in other circumstances. This technique has been frequently used for air pollution exposure assessment (Wu et al., 2005; Valero et al., 2009). The advantage of regression models is their relative simplicity and that direct measured data are employed. However, such models are only applicable for the specific data and circumstances under which they were derived, since no account is taken of the underlying physical principles. In addition, regression models are generally based on measurements over short time periods which are then extrapolated to model long-term averages. Consequently, assumptions and high levels of uncertainty can be introduced (Milner et al., 2010).

4.3.2 Mass-Balance Approaches

Instead of using measurements, mass-balance techniques model the flow of air and pollutants through a simple system of one or two compartments that are connected to the outdoor air. Concentrations of pollution can then be linked to time series occupant activity data to calculate personal exposure (Nazaroff and Cass, 1986). The concentration in each compartment is determined as a function of key building parameters: the outdoor concentration, indoor source strengths and the physical properties of the air pollutants. Given that the rate of airflow entering and leaving each compartment must be equal, the concentration, C, in that a compartment may be represented by an ordinary differential equation of the form:

$$dC/dt = S - LC \qquad \text{Equation 5}$$

Where dC/dt is the rate of change of the concentration of the pollutant (in this instance $PM_{2.5}$), S is the sum of all sources and L is the sum of all sinks (Milner et al., 2011). The models require data on outdoor air pollution concentrations, the building's air change rate and indoor source emissions and sinks. Neither of these two approaches however capture the level of complexity required to accurately estimate indoor $PM_{2.5}$ concentrations on a large scale, or the complex and variable housing stock required for the purposes of this study.

4.3.3 Multizone Models

Based on the same principles as mass-balance approaches, these more complex models have multiple linked compartments allowing for detailed building geometry, contaminant, and airflow component data. Consequently, users can specify complex building and component configurations, exact room dimensions, building orientations, ventilation components (e.g. large openings), occupant activities with time schedules, cracks, extract fans, trickle vents and HVAC or MVHR systems (Milner et al., 2010). External transient weather files enable spatial and temporal fluctuations in building air change rate and pollutant transport due to, for example, wind pressures or stack effects. Multizone models enable multiple indoor source types and characteristics, with detailed scheduling of source-based activity allowing changes in source emission rates over time. Models can be easily adjusted to simulate changes due to the application of a range of energy efficiency measures, ventilations strategies linked to occupant behaviour and varying external conditions (Dimitroulopoulou et al., 2001; Glytsos et al., 2004). Such models provide a closer representation of real-world situations, without the complexity of Computational Fluid Dynamics (CFD) analysis. In addition, internal temperature estimates from energy models such as EnergyPlus (US-DOE, 2013) may be input into the models to give a more complete picture of the impacts on concentrations of $PM_{2.5}$ due to dynamic temperature effects. Alternatively, coupled dynamic temperature and contaminant models have been developed in EnergyPlus (not publically released at the time beginning of this study) and a version of CONTAM has been coupled to TRNSYS - an external energy analysis programme- (NIST, 2015). The latter however, was prohibitively expensive to be used in this study. Multizone modelling using validated programmes provides a useful means of studying a large number of building variants and allows the rapid calculation of the movement of contaminants within buildings under dynamic and changing conditions.

4.3.4 Computational Fluid Dynamics (CFD)

Such analysis is primarily used by building environmental designers to calculate the temperature, velocity and other fluid properties through a 3-D domain, although pollutant transport can also be modelled. Using conventional CFD packages for building airflow and pollutant analysis can be a time consuming task requiring very careful attention to setting up the correct geometry and boundary conditions (Nielsen, 2004). CFD is not considered appropriate for generic population exposure modelling but may be useful as a means for examining the determinants of personal exposure at finer spatial and temporal scales (Milner et al., 2010; Gilkeson et al., 2014), or individual ventilation components/systems Capetillo et al., 2015). It was therefore considered unsuitable for modelling the wide range of housing variants and stock/population level modelling investigated in this research. However, as rapid improvements in CFD capabilities are underway with increases in computer power, it is possible that future research could use CFD analysis, building upon the work presented in this thesis.

4.4 Selected Modelling Programmes

Due to the nature of this study, its scope, objectives and the level of complexity required, CONTAMv2.4c was chosen as the primary tool for this study. CONTAM is a freely available extensively validated multi-zone airflow and pollutant transport simulation tool that enables the construction of various dwelling types with inputs from numerous disparate data sources, which can be adjusted to gauge their impact on $PM_{2.5}$ concentrations (Emmerich, 2001, Walton and Dols, 2005). By running multiple scenarios, where individual variables are systematically altered (parametric analysis), key inputs that affect personal exposure to $PM_{2.5}$ can be identified as well as those which have negligible or no impact and do not require such accurate specification. In addition, in the later part of the project, EnergyPlus (a validated energy analysis and thermal load simulation program) (US-DOE (2013), was utilised following the implementation of the Generic Contaminant Model in Version 7.2, which is able to model coupled dynamic thermal simulation and $PM_{2.5}$ concentrations. Prior to this, dynamic temperature profiles were imported from EnergyPlus into CONTAM. Where there are differences between the programmes and the inputs listed under section 3.5 they are covered in the individual chapters in which they occur (Chapters 5,6&7).

4.4.1 Multizonal Models: Assumptions and Limitations

Each building modelled within CONTAM and EnergyPlus was created as a series of nodes which represent zones with airflow elements such as cracks, windows, doors and ducts. The resulting simultaneous non-linear equations determine the flow through the building with the ability to model whole building infiltration and ventilation rates. By using this network model, transient weather files with variable wind speed and direction were employed to generate more realistic scenarios than those achieved with fixed values. In addition, buoyancy and stack-effect impacts are produced using variable internal to external temperature profiles. It was then possible to predict consequent $PM_{2.5}$ concentrations which were dispersed by these airflows, including absorption by and deposition to building surfaces and re-suspension. $PM_{2.5}$ from both internal and external sources could be modelled in each zone, as well as the transport between zones. In addition, a range of purpose provided ventilation systems with various flow rates and other energy efficiency measures could be constructed, enabling the impact of climate change mitigation interventions to be evaluated. Time/event based schedules for activities within domestic properties and components give an indication of occupant behaviour influences. Post processing of results allows the effect of this typical occupant behaviour on $PM_{2.5}$ exposures to be estimated.

As with all multizone models, there is an inbuilt assumption that the air and therefore the contaminant is uniformly mixed within each zone, which does not allow for spatial variation within zones, whereas in reality any individual close to a source may be exposed to a higher concentration. Conversely, other members of the household may receive lower than estimated exposure. A CFD component (add-on) has more recently been developed for CONTAM as well as a method for two dimensional modelling for zones such as hallways in order to address these issues (Wang et al., 2010). However, given the complexities of applying these to a large number of buildings in the modelled building stock and the additional computer processing required, these have not been used in this study. Instead, post-processing of the zone concentrations in CONTAM was used to quantify exposures for different occupants based on their movement around the building over a year rather than simply giving room average concentrations). Sensitivity analysis (section 5.1.7) is used to calculate uncertainties in the programme outputs, which are caused by those in both the computational processes and data used as input variables (Dutton, et al., 2008).

4.5 Model Inputs and their Sources

4.5.1 Introduction

This sub-section describes the background methodology and assumptions in developing the models, including:

4.5.2 Building stock databases

4.5.3 Building stock modelling; housing archetypes

4.5.4 Ventilation components

4.5.5 Outdoor $PM_{2.5}$

Greater detail is provided in the relevant chapters, specific to the research question investigated. The choice of input variables used in this study was mainly determined by the empirical data available in the literature. Where these provided a range of values, the largest studies, using the most effective monitoring equipment and/or those most relevant to the UK were used, with sensitivity analysis enabling investigation of the distribution of data outputs. The input variables used in the base-case model represent the mean of a distribution or range of values, with the exception of external $PM_{2.5}$ concentrations and weather data where ratified (that is, confirmed after additional analysis) data from monitoring stations were used. Wherever direct empirical data was available (as opposed to modelled values), input variables use these values in order to improve the accuracy of the models. The key input variables used with the programme are listed under section 4.5.4 onwards.

4.5.2 Building Stock Databases

In England, the housing stock has been formed over a long period of time, with a variety of different materials and construction techniques being used. Any housing stock model used for calculating indoor concentrations of $PM_{2.5}$ will be complex and as such a range of assumptions are necessary in order to develop a model representative of the English stock due to the variations in the physical form and construction of dwellings. There are a variety of building stock databases in the literature which are primarily employed in physics-based bottom-up modelling where they are used to construct archetypes for modelling from a range of disaggregated components. A review by Kavgic et al. (2010), highlighted 9 different model types with spatial resolutions (level of disaggregation) ranging from 2-20,000 dwelling types, depending on the various definitions of 'uniqueness'. In addition to these, the National Energy Efficiency Data-framework (NEED) was used by Wyatt, (2013) to investigate the physical and socioeconomic drivers of energy consumption in England; the British Household Panel Survey (BHPS) is a continuous (longitudinal) survey, but lacks sufficient detail for this investigation (BHPS, 2013). The English housing survey (EHS), commissioned by the UK Government, is a longitudinal national survey collecting information about people's housing circumstances and the condition of housing in England. It has information that enables quantification of the distribution of housing stock types at the regional level. (EHS, 2012). The EHS provides data on key housing stock characteristics (including age, type and size) and households (age, tenure, occupants, income, vulnerability) and are based on physical surveys and occupant interviews. The surveyed 'dwelling sample' of properties where physical inspections were carried out contains 16,150 occupied or vacant dwellings of the housing stock of 22.5 million dwellings in England. The EHS, currently the most comprehensive stock data base available and is considered fit for purpose in term of the requirements of this study as it contains the breadth of housing characteristics and the majority of other variables needed for modelling inputs and stock level construction. The current occurrence of energy efficiency and ventilation interventions in the English stock for the locations studied can be supplemented with information from other databases (e.g. HECA, 2013; HEED, 2014). Consequently, the profile of current English dwellings can be modelled and the range and distribution of existing properties that are available for energy efficiency and ventilation interventions clarified.

4.5.3 Building Stock Modelling: Housing Archetypes

Building stock modelling is the process of representing the existing essential building features and characteristics (as relevant to research question, e.g. dimensions, materials, ventilation strategies, building permeability) in a specific location to be reproduced using a software programme (Oikonomou et al., 2012). Building stock modelling can be divided into (i) top-down and (ii) bottom-up approaches. Top down models are constructed using aggregated data (for example historic energy consumption data), however they lack the level of constructional detail required for indoor pollutant modelling. Bottom-up models allow the combination of a variety of disparate components, which can be constructed in a building physics based model to estimate the impact of each component. Numerous bottom up models exist, capable of dealing with different ranges of data (Kavgic et al., 2010). These models can be easily adapted to explore future scenarios including the impact of different ventilation strategies and energy efficiency interventions in order to identify the effect of emission reduction schemes on indoor environmental quality, (Kavgic et al., 2010) – in the case of this study the concentrations of $PM_{2.5}$. The use of building stock modelling enables the aims of this research to be achieved without the need for further expensive and time consuming empirical monitoring and is the only method available for investigating possible future scenarios (Johnson, 2001).

For the purposes of this research, one of the crucial aspects of building stock modelling is the identification of archetypes (i.e. various dwelling types with different layouts, geometries and sizes) which are 'representative' of the most prevalent housing types found in the particular building stock that one wishes to model. Two approaches were taken in building stock modelling for this study, one building upon the other as the investigation progressed. In the preliminary analysis seen in Chapter 5 the two dwelling types used in a study published in the Lancet (Wilkinson et al., 2009), were utilised and adapted in order to carry out investigations into changes in indoor $PM_{2.5}$ concentrations in London's housing stock. Those were chosen as ONS data show that the London stock is divided in 50% flats and 50% houses (ONS, 2011). Multiple (variants) of the baseline models were constructed by applying a range of ventilation strategies complying with current Building Regulations (ADF 2010), adding multiple $PM_{2.5}$ indoor sources and conducting occupant behavioural and sensitivity analysis in order to quantify personal $PM_{2.5}$ exposure profiles for different residents. The results from this phase of the study, which are covered in Chapter 5 were published in a peer-reviewed journal (Shrubsole et al., 2012) see Appendix H.

Following this preliminary work, further archetypes were added to more accurately reflect the major domestic geometries in England and were selected for full analysis in order to obtain a broader examination of the distribution of $PM_{2.5}$ exposures. Nine were obtained from the LUCID project (Oikonomou et al., 2012) including three flat and six house types. These archetypes and their many variants were derived from the English Housing Survey (EHS, 2012). The criteria used for archetype selection for simulation involved choosing the most frequently occurring properties, with nine archetypes representing over 76% of the properties in the investigated area. Those archetypes not used had many characteristics similar to those modelled (Mavrogianni et al., 2012; Oikonomou et al., 2012). One further built form was added to complete the range; a detached dwelling, called House 7 from the Lancet study (Wilkinson et al., 2009) that represented the built form with the next highest representation seen in the English Housing Survey (EHS, 2012), but was absent from the other studies. By including House 7, it is assumed that these archetypes are broadly representative of the English domestic stock. The EHS also gives details of the frequency of their individual occurrence in the areas studied. This more comprehensive stock model was used to investigate the impact of locational factors (Chapter 6). Table 4.1 gives a generic description of the set of archetypes (see appendix D for full details of each archetype).

Table 4.1 Generic description of CONTAM Stock archetypes used in Chapters 5 and 8

Model Reference	Outline built form
Flat 1	1 bed layout 1
Flat 2	1 bed layout 2
Flat 3	1 bed layout 3
House 1	3 bed terrace
House 2	2 bed terrace
House 3	2 bed semi
House 4	5 bed terrace
House 5	3 bed bungalow
House 6	3 bed terrace above shop
House 7	3 bed detached

In Chapter 6, the archetypes are incorporated within the SCRIBE* Tool to aid investigations of the influences of location on indoor $PM_{2.5}$ concentrations. In order to consider the impact of energy efficiency strategies on different income groups and tenures, the baseline archetypes and their variants were reconstructed using EnergyPlus, an energy analysis and thermal load simulation program with a multizone airflow and contaminant transport analysis component (US-DOE, 2013). Using these two programmes also served to aid the investigation of sensitivity analysis and model uncertainty seen in Chapter 7 and Appendix E. Table 4.2 lists the sources for baseline geometries and the modelling programmes used. As the study progressed (chapters 5-7), the level of complexity and the range of modelled geometries and their variants also increased. This was partly driven by the scope of the particular project behind the work, but also as better and more comprehensive data sources became available.

Table 4.2 The various baseline archetypes and software used for thesis investigations

Chapter	Archetype	Software
5	Wilkinson et al, 2009	CONTAM
6	Wilkinson et al, 2009; Oikonomou et al., 2012	CONTAM (within SCRIBE*)
7	Oikonomou et al., 2012; Awesome, 2013	EnergyPlus
7	Wilkinson et al, 2009; Oikonomou et al., 2012; Awesome, 2013	CONTAM, EnergyPlus
8	Wilkinson et al, 2009; Oikonomou et al., 2012;	CONTAM, EnergyPlus

*Discussed in detail in Chapter 6

All of these investigations lead to research publications, which are detailed within the relevant chapters and Appendix H. An example of a flat is shown in Figure 4.2 with construction dimensions in Table 4.3. Full details of all dwelling archetypes are provided in Appendix C.

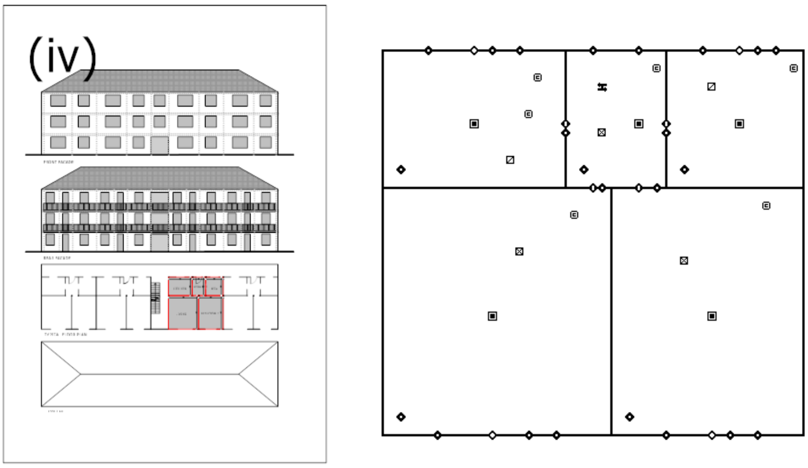

Figure 4.2 Flat 1: 1 bed (archetype type- IV) dwelling reference H04 & H07 (Oikonomou *et al.*, 2012).

Table 4.3 Construction Dimensions Flat 1

Flat 1 (1 bedroom) – Construction Dimensions						
Footprint				54.60 m^2		
Number of floors				1		
Floor to ceiling height				2.60 m		
Envelope area				180.96 m^2		
Permeable envelope				40.56 m^2		
Room	Kitchen	Living	Bed 1	Entrance	Bathroom	Total
Floor area m^2	8.50	19.35	15.75	4.75	6.25	54.60
Volume m^3	22.10	50.31	40.95	12.35	16.25	141.96

4.5.4 Ventilation Components

Guidance in Approved Document Part F (ADF 2010) (HM Government, 2010) has been used in the implementation of all purpose-provided ventilation components in the models, with all systems assumed to be functioning correctly and no allowance made for mechanical failure, blockages or deterioration with time. However, in addition to window opening, all buildings have a measure of uncontrolled ventilation due to defects in the structural components. Using a blower-door test and with knowledge of the building dimensions, building permeability can be calculated (BS EN13829, 2001). For this study, test data from Stephen (1998 and 2000) is used to produce a distribution of permeabilities for the English housing stock. The prevalence of controlled ventilation systems, both active and passive (extractors and trickle vents) within the models are based on two factors; 1) The percentage of the English stock built subsequent to Building Regulation requirement for these measures and 2) an additional percentage of properties built prior to the regulations but having these measures based on their occurrence seen in data from the Warm Front Study, prior to any interventions being added (Warm Front, 2011). For extract fans Variables finkxtwk and finbxtwk in EHS (Amenity) show the presence of working extract fans in the kitchen and bathroom. For trickle vents, their presence was determined as follows: All post-1990 dwellings are assumed to have trickle vents as these were an approved system in the 1990 building regulations, to meet ventilation requirements. The Warm Front survey has identified that approximately 5% of pre-1990 dwellings have at least 8 trickle vents installed, an approximation for meeting Approved Document F (1990) requirements. A 5% sample of pre-1990 dwellings in the modelling are designated as has having operational trickle vents. All other pre-1990 dwellings are assumed to have no trickle vents or extract fans.

4.5.4.1 Building Permeability

Dwelling permeability in the models is provided by adventitious openings (infiltration component), which represent cracks between external building components and in the exposed building facade. These openings are placed in the external walls, floors and roofs, with gap size proportional to facade area. The permeability of the exposed façades in the conditioned part of the building envelope is modelled using two cracks placed at the top and bottom of the wall (Orme & Leksmono, 2002). Jones et al., 2012 suggest that this method may overestimate the buoyancy impact, as the distance between the paths is at maximum. For this reason, crack positions and relative distances are addressed within the sensitivity analysis seen in Chapter 5.

The airflow through each crack is modelled using the following two-way flow power-law equation with a flow exponent (n) assumed to be 0.66 for gaps and cracks and a flow coefficient proportional to the permeability at 50Pa multiplied by the area of the exposed façade (Fang and Persily, 1995).

$F=PS/3600$ Equation 6

Where:

F = Flow rate in m^3/s @50Pa
P= Permeability $m^3/m^2/hr$ @50Pa
S= Total external exposed surface area (m^2)

The calculation can be transposed to enable calculation of the flow coefficient (C)

$C=F / (\Delta P)$ ^n Equation 7

Where:

$\Delta P = 50Pa$
n is the flow exponent indicating the degree of turbulence (0-1).

An n value of 0.5 represents fully turbulent flow and 1.0 represents fully laminar flow. The typical n value for a whole building is 0.66 (Fang and Persily, 1995). This gives the flow coefficient (C) for a particular external facade of area (S), which includes any windows within this area. If there is more than one gap/crack, then this figure is divided by the number of gaps/cracks per external exposed facade to give a coefficient (c) for each opening. An example of flow coefficient (C) calculation for adventitious openings in the external wall of the living room in Flat 1 is shown in Table 4.4

Table 4.4 Example of permeability coefficient calculation for Flat 1 external living room wall

Flat 1 External living room wall								
Item	Range of Values							
Permeability (m^3/ hr/ m^2 @50Pa)	3	5	7	10	15	20	25	30
Height (m)	2.6	2.6	2.6	2.6	2.6	2.6	2.6	2.6
Length (m)	4.3	4.3	4.3	4.3	4.3	4.3	4.3	4.3
Surface Area (m^2)	11.18	11.18	11.18	11.18	11.18	11.18	11.18	11.18
Flow (m^3/hr @50Pa)	33.5	55.9	78.26	111.8	167.7	223.6	279.5	335.4
Flow (m^3/s @50Pa)	0.0093	0.0155	0.0217	0.0310	0.0465	0.0621	0.0776	0.0931
C	0.0007	0.0011	0.0016	0.0023	0.0035	0.0046	0.0058	0.0070
c, for each of the 2 cracks per wall	0.0003	0.0005	0.0008	0.0011	0.0017	0.0023	0.0029	0.0035

Equation 7 was used to produce flow coefficients for eight levels of permeability: 3, 5, 7, 10, 15, 20, 25 and 30 $m^3m^{-2}hr^{-1}$ at 50Pa. These values reflect the observed range of permeabilities in the UK domestic stock (Stephen, 1998) and are also assumed to be broadly representative of the individual locations investigated in this thesis. For each of these permeabilities, there is distribution of the English stock that falls into that category. These percentages have been applied when calculating the representative $PM_{2.5}$ concentrations for each stock archetype, variant and permeability.

Table 4.5 Distribution of permeabilities in the English housing stock (Stephen, 1998; 2000).

Distribution	1	10	10	21	35	19	3	1
Permeability (m3/m2/hr @50Pa)	3	5	7	10	15	20	25	30

This has been confirmed by later studies showing little change even among some new build properties (Stephen, 2000; Grigg, 2004). This process was conducted in each room in all geometries with openings placed on all external walls, floors and lofts with gap/crack sizes proportional to the external façade area in question.

Internal walls are considered impermeable within CONTAM, however this is not the case in EnergyPlus where the airflow network requires cracks on internal surfaces including walls, floors, and ceilings. This issue is addressed in Chapter 7 and is the subject of a research paper to which the author contributed

(Jones et al., 2013). Doors when shut, have gaps representing 1% of the open area between door and frame as the estimation of the gap based on standard door to frame clearances. Internal doors (excluding storage) are always modelled open except during activities such as cooking, sleeping and bathroom use. For dwellings incorporating an MVHR system, an undercut of minimum area 7,600mm² in all internal doors above the floor finish (10mm gap on a standard door) is modelled to allow airflow for correct functioning in line with Approved Document F, 2010.

4.5.4.2 Purpose Provided Ventilation (PPV)

The Building Regulations, specifically Approved Document L1A: Conservation of fuel and power in new dwellings and Approved Document L1B: Conservation of fuel and power in existing dwellings (AD, 2010), set out the requirements for improvements in the Target Emission Rate (TER). It is unlikely that buildings will be able to achieve the overall targets for energy improvement without significantly improving air tightness. Consequently, changes in Approved Document F: Means of ventilation, are needed to ensure adequate purpose provided ventilation (PPV) is delivered to supply a healthy level of air changes in domestic buildings (AD, 2010). The ventilation strategy for dwellings includes three main elements that can be delivered either by natural or mechanical systems, or a combination of both including:

- Extract ventilation - mechanical extract for rooms with water vapour or pollutants.
- Whole building ventilation - providing fresh air and dispersal of pollutants through air exchange via background (trickle) ventilators.
- Purge ventilation - to remove large amounts of pollutants and vapour and to be used to improve thermal comfort.

For this reason, extract fans, trickle ventilators, MVHR systems, air bricks and eaves ventilators within the models were specified to comply with the relevant Building Regulations. The sizes of individual components were then matched to those currently manufactured in the UK, as described below.

Extract fans (where present) use intermittent minimum rates as per Table 5.1a of ADF 2010 (HM Government, 2010) during cooking activities and bathroom/toilet occupation as shown in Table 4.6. Based on the absence of suitable data, reasonable assumptions were made regarding occupant behaviour with fixed periods and durations for all domestic activities assumed.

Table 4.6 Intermittent extract ventilation rates and schedules

Intermittent extract ventilation rates and schedules			
Day	Room	Extract Rate	Schedule
Weekday	Kitchen	60 l/s	07:30 to 08:30, 18:00 to 19:30
	Bathroom	15 l/s	07:00 to 08:00, 19:30 to 20:30, 21:30 to 22:00
	Toilet	6 l/s	07:00 to 08:00, 19:30 to 20:30, 21:30 to 22:00
Weekend	Kitchen	60 l/s	08:30 to 09:30, 12:00 to 12:30, 18:00 to 19:30
	Bathroom	15 l/s	08:00 to 09:00, 19:30 to 20:30, 21:30 to 22:00
	Toilet	6 l/s	08:00 to 09:00, 19:30 to 20:30, 21:30 to 22:00

Trickle ventilators use the minimum background ventilation rates from Table 5.2b of ADF 2010 for properties with permeabilities ≥5m3/m2/hr@50Pa. An example of the minimum area calculations for the CONTAM archetypes are show in Table 4.7. All trickle vents are modelled using the power law Q=C (ΔP) ^n, with a leakage area per unit length of 0.001m^2/m; a pressure drop of 1 Pa; a discharge coefficient of 0.62 and a flow exponent of 0.5 (the values suggested for larger openings by Walton and Dols, 2005). In all properties, the minimum requirements of 5000mm^2 (equivalent area[1]) per habitable room and 2500mm2 (equivalent area) in wet rooms were modelled. Further vents were placed proportionately amongst all rooms, to ensure total building ventilation compliance with ADF 2010 for all geometries.

[1] Equivalent area is used instead of free area for the sizing of trickle ventilators. Free area is the physical size of the aperture of the ventilator but may not accurately reflect the air flow performance. The more complicated and/or contorted the air flow passages in a ventilator, the less air will flow through it. So, two different ventilators with the same free area will not necessarily have the same air flow performance. A European Standard, BS EN 13141-1:2004 (Clause 4), includes a method of measuring the equivalent area of background ventilator openings. Trickle ventilators can be modelled as an open orifice, thereby achieving equivalent area requirements of ADF, 2010.

Table 4.7 Example of minimum equivalent area (ADF 2010) for trickle vents in the CONTAM archetypes

Minimum equivalent area for trickle vents (ADF 2010)	
Geometry	mm^2
Flat 1	25000
Flat 2	25000
Flat 3	25000
House 1	75000
House 2	60000
House 3	45000
House 4	105000
House 5	40000
House 6	75000
House 7	45000

Airbricks/Vents in those dwellings having underfloor areas, models were constructed to conform to BS5250 section 8.5.3, (HM Government, 2011a), which contain two methods of calculation for minimum ventilation; either $500mm^2$ of ventilation per m^2 of the cellar/underfloor area *or* $1500mm^2$ of ventilation per linear m of the total external wall length. Compliance requires that the greater result from the two calculations be used. Results for CONTAM geometries are shown in Tables 4.8 and 4.9, with the highest results highlighted in grey. In order to comply with the Standard, the numbers of airbricks required was calculated using those sizes of units currently available from UK manufacturers.

Table 4.8 Example of calculation of number of airbricks required BS5250 (m^2 cellar rule)

Number of airbricks required : Geometries		
Geometry	m^2 required: $500mm^2/m^2$ rule	number of airbricks
House 1	0.04	7
House 2	0.03	6
House 3	0.02	
House 4	0.04	7
House 5	0.04	
House 6	0.04	7
House 7	0.02	

Table 4.9 Example of calculation of number of airbricks required BS5250 (linear external wall rule)

Number of airbricks required: Geometries		
Geometry	m^2 required: $1500mm^2/m$ rule	number of airbricks
House 1	0.03	
House 2	0.02	
House 3	0.03	5
House 4	0.02	
House 5	0.06	10
House 6	0.02	
House 7	0.04	7

Loft Vents are modelled to comply with BS5250 section 8.4.2.2.3.2.
(HM. Government, 2011a). The open width of the ventilator is 10mm x eaves length and the roof pitch is assumed to be >15°.

4.5.5 Outdoor $PM_{2.5}$

For current (2010) outdoor $PM_{2.5}$ concentration, data was calculated by analysing the empirical data downloaded from urban background stations from the Automatic Urban and Rural (AURN) and local authority run networks where this data was available in the areas being studied. Where this was not available, concentrations are based on data from the DEFRA mapping project (DEFRA, 2013). For future $PM_{2.5}$ concentrations, DEFRA data is used up to 2030 in both locations. For 2050, in the absence of further data, a linear trend based on 2010-2030 data is assumed. The modelling enables differentiation between $PM_{2.5}$ from indoor and outdoor sources due to the ongoing uncertainty regarding the toxicity of particles generated indoors (Wilkinson et al, 2009).

4.6 Summary and Implications for Research

This chapter has sought to explain in general terms, the modelling techniques and processes adopted and the process for selection of the various inputs required in order to investigate the questions posed by this thesis. Some specific aspects are slightly different depending on whether CONTAM (chapter 5&6) or EnergyPlus software (chapter 7) is utilised. These are dealt with within the relevant chapters. The primary function of this chapter is to enable easier understanding of the research that follows. In the next chapter an analysis of indoor concentrations of $PM_{2.5}$ using CONTAM in the current and a possible future London stock, subject to a particular energy efficiency and ventilation strategy is detailed.

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Chapter 5

Predicted Changes in Indoor $PM_{2.5}$ in London's Domestic Stock, due to Energy Efficiency Retrofits: An Idealised Case

Introduction

This chapter describes how CONTAM was employed to create simulations to predict indoor exposure to $PM_{2.5}$ from both indoor and outdoor sources in the Greater London Authority (GLA). It considers London dwellings in both the present day (2010) housing stock and the same stock following specific energy efficient refurbishments in order to meet greenhouse gas emissions reduction targets for 2050. London was chosen as the starting point for investigations due to its mature and well developed building stock along with easily available and long term data sets for inputs required for the CONTAM modelling. It represents over 13% of the UK population and experiences poor air quality, having failed to achieve compliance with EU standards for $PM_{2.5}$ and other airborne pollutants in recent years. In addition, as this work initially builds on previous work carried out by Wilkinson et al., 2009, which characterised the London stock, it proved invaluable for checking and testing the newly developed models and the energy efficiency and ventilation interventions. The interventions modelled are those that would contribute to the achievement of these targets by reducing the permeability of all dwellings to $3m^3m^{-2}hr^{-1}$ at 50Pa, combined with the introduction of mechanical ventilation and heat recovery (MVHR) systems in all dwellings. Sensitivity analysis is carried out to examine the sensitivity of the results to model inputs and assumptions. In addition, the location of properties in relation to external pollutant source (in this instance vehicle activity) impacts the concentration of the external component of $PM_{2.5}$ entering properties and is investigated using the Operational Street Pollution Model (OSPM, 2010). Occupant behaviour is studied to ascertain any variations in $PM_{2.5}$ concentrations experienced by different members of a household. For the exposures, modelled health impact assessments (HIA) are conducted to quantify the health impacts of changes in indoor $PM_{2.5}$ exposure.

The investigation in this chapter resulted in a peer reviewed research paper (Shrubsole et al., 2012) as well as two related conference papers and input into other papers as detailed under 'thesis associated publications' beginning at page 14.

5.1 Methodological Approaches

5.1.1 Basic Building Stock Modelling

The indoor study was based on the application of CONTAM (Emmerich, 2001) to predict concentrations of particles with maximum aerodynamic diameter less than or equal to 2.5 microns ($PM_{2.5}$) from both indoor and outdoor sources, in specific zones/rooms of dwellings. This model develops previously published methods of exposure characterization to $PM_{2.5}$ (Wilkinson *et al*., 2009). It includes a more detailed approach to modelling including the effect of ventilation systems (specifically MVHR) within dwellings, multiple $PM_{2.5}$ sources, occupant behaviour (such as cooking times, washing, cleaning, sleeping etc.) location impacts and sensitivity analysis. It also uses a more comprehensive and broader range of empirical data sources as model inputs. An outline of the modelling approach is shown in Figure 5.1.

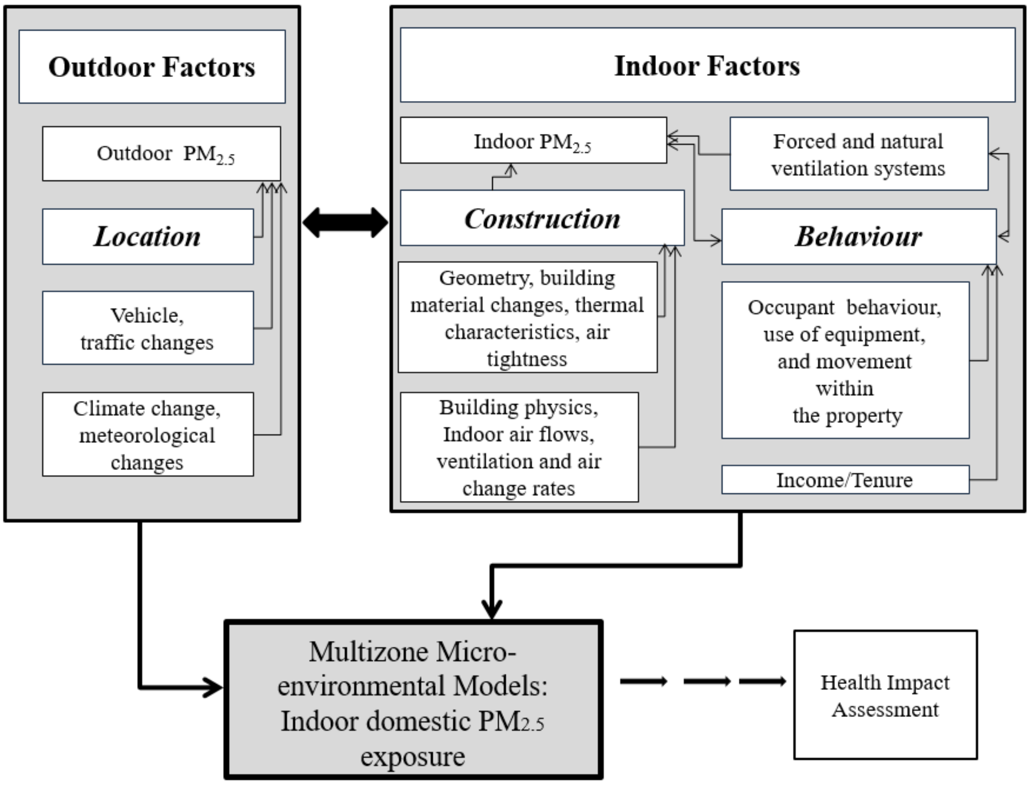


Figure 5.1 Range of factors affecting modelling of indoor $PM_{2.5}$ exposure and preparation of data for future input to heath impact assessment.

The modelling was carried out to simulate indoor $PM_{2.5}$ concentrations in dwellings using dwelling characteristics selected to be broadly representative of the London housing stock, with Office of National Statistics (ONS) data showing a 50/50 split between houses and flats in the capital in 2010 (ONS, 2011a). The two archetypes include a Flat (Figure 5.2 and Table 5.2) and a House (Figure 5.3 and Table 5.3), which were developed as typical archetypes used in a study on the investigation of ventilation effectiveness in support of Part F of the Building Regulations (FMNectar, 2007) and subsequently used in the Lancet study considering the impacts of climate change mitigation strategies in dwellings (Wilkinson et al., 2009).

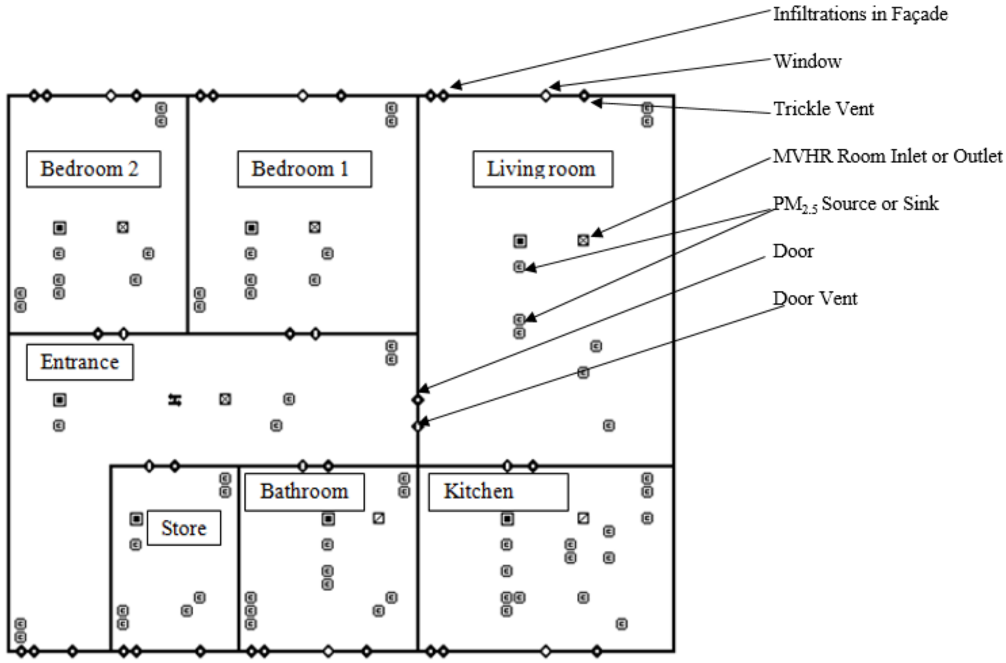
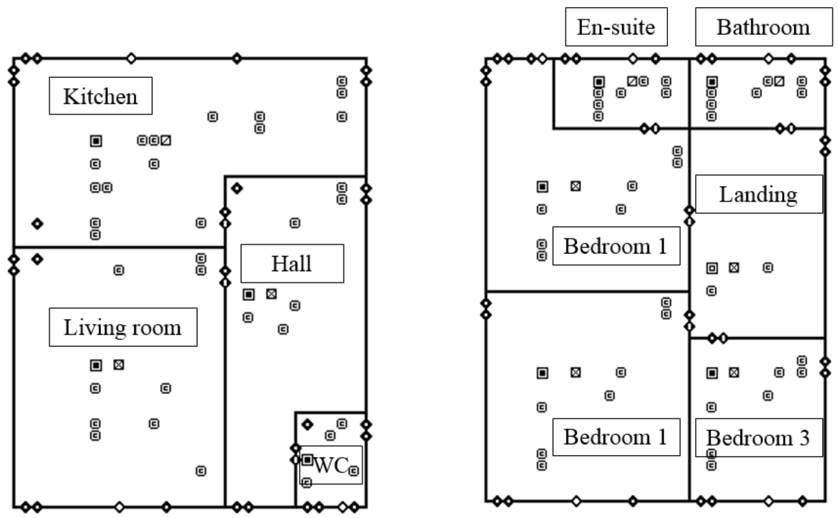


Figure 5.2 Example plan of the simulated Flat in CONTAM with modelled rooms, pollutant sources/sinks, ventilation systems, windows, doors and adventitious infiltrations.

Table 5.1 Construction Dimensions: Flat

Flat: Dimensions								
Foot print	45.00m²							
Number of floors	1							
Floor to ceiling height	2.4 m							
Envelope area	64.8 m²							
Permeable envelope (internally impermeable)	36.0 m²							
Room	Hall	Store	Bathroom	Kitchen	Living	Bedroom1	Bedroom 2	Total
Floor area m²	7.8	2.7	3.6	5.0	11.8	8.1	6.0	45.0
Volume m³	18.7	6.5	8.6	12.0	28.3	19.4	14.4	107.9

The flat (Figure 5.2) was modelled to be on the ground floor, with no adjustments for either changes in wind speed or $PM_{2.5}$ concentrations with height. This investigation is carried out separately under section 5.1.6.



Ground Floor Plan First Floor Plan

Figure 5.3 Example plan of the simulated House in CONTAM with modelled rooms, pollutant sources/sinks, ventilation systems, windows, doors and adventitious infiltrations. The CONTAM models also have an underfloor plan (where appropriate) and a roof plan (not shown here).

Table 5.2 Construction Dimensions: House

House: Dimensions											
Footprint	48.00 m^2										
Number of floors	2										
Floor to ceiling height	2.40m										
Envelope area	230.4 m^2										
Permeable envelope	230.4 m^2										
Room	Kit	Liv	Bed 1	Bed 2	Bed 3	Land	WC	Hall	En-suite	Bath room	Total
Floor area m^2	18.60	16.20	12.60	12.60	6.00	9.60	1.80	11.40	3.60	3.60	96.00
Volume m^3	44.60	38.90	30.30	30.20	14.40	23.00	4.30	27.40	8.60	8.60	230.40

Simulations using these archetypes were run to investigate the influence of combinations of key parameters: dwelling type, geometry, ventilation system and permeability (Table 5.4). Consequently, a range of variants were developed to represent the current and possible future London domestic stocks from the two baseline archetypes. which included the range of permeability seen in the English domestic stock, winter and summer versions to represent different window opening behaviour and the range of ventilation components seen in the current stock, leading to a total of 135 individual models for the current and future GLA domestic housing.

Table 5.3 Summary of key dwelling variants and external environment characteristics for the current and possible future London housing stock.

Parameter	Current (2010) housing stock	Stock under future (2050) scenario
Ventilation regimes	(1) infiltration and purge ventilation only: (2) infiltration, trickle ventilators, extraction fans and periodic purge* ventilation;	ventilated via MVHR systems with background /boost modes and filters that remove 80% of $PM_{2.5}$
Permeability	3, 5, 7, 10, 15, 20, 25, 30 $m^3m^{-2}hr^{-1}$ at 50 Pa	$3m^{-3}m^{-2}hr^{-1}$ at 50Pa + MVHR systems with filters removing 80% of $PM_{2.5}$.
Outdoor $PM_{2.5}$	13 $\mu g.m^{-3}$	9 $\mu g.m^{-3}$

*periodic purge represents the opening of windows for a limited period following events such as bathroom use or cooking, where no other means of ventilation (e.g. extract fans) exists.

5.1.2 Data Inputs: Emissions and Depositions Rates and Scheduling

The key inputs to the CONTAM models are summarized in Table 5.4. The emission rate of $PM_{2.5}$ from cooking was assumed to be 1.6 $mg.min^{-1}$ $\pm$ 0.6 $mg.min^{-1}$ based on 4.1 $mg.min^{-1}$ $\pm$ 1.6 $mg.min^{-1}$ of inhalable PM_{10} of which 40% is the finer fraction of $PM_{2.5}$ having a $PM_{2.5}$ deposition rate of $0.39hr^{-1}$ (figures derived from the large scale PTEAM study (Ozkaynak et al., 1996)). However, published emission rates for cooking vary greatly, depending on food type, cooking method, appliance and method of measurement (He et al., 2004; Olson and Burke, 2006). Consequently, for sensitivity analysis, models were run using estimates to $\pm$ 2.33 standard deviations. For resuspension from dusting and vacuuming or sweeping, and in the absence of better data sources, the initial surface loadings reported in Ozkaynak et al. (1996); He et al. (2004) and Afshari et al. (2005) were assumed to be applicable to the London stock.

Deposition rates for $PM_{2.5}$ are calculated from the deposition velocity multiplied by (A/V) where A is the surface area and V the volume of a given space. The literature however, gives a variety of preferences as to what values for “A” should be used, with some authors linking this to particular $PM_{2.5}$ fractions and surfaces and others not; for example: Thatcher and Layton,1995; Fogh et al., 1997; Thornburg *et al*., 2001. Whilst it is acknowledged that the various particle sizes within $PM_{2.5}$ behave differently (Thornburg et al., 2001), for the purpose of this study, $PM_{2.5}$ has been considered as a whole and not broken down into sub-components or particle sizes in line with similar modelling studies (Dimitroulopoulou et al., 2006; Wilkinson et al., 2009). The PTEAM deposition rate for $PM_{2.5}$ of $0.39hr^{-1}$ is an average for all sub 2.5μm particles and was used in the CONTAM modelling to calculate a deposition velocity based on the geometries from Wilkinson, 2009 and run to compare the I/O ratios obtained against monitored data to confirm its suitability for the London stock models by the use of equation 3 $Dr = Dv\ (A/V)$ see section 4.1.4.

All model scenarios were run with and without a source of $PM_{2.5}$ from tobacco smoke in order to calculate the impact of $PM_{2.5}$ from smoking on non-smoking members of the household. In dwellings occupied by a smoker and using data from the Office of National Statistics data (ONS, 2000), it was assumed 1 cigarette was smoked per waking hour; the scheduling used acknowledges that smoking occurs both outdoors and also indoors in the kitchen and living room (HSCIC, 2012). A no-internal $PM_{2.5}$ source scenario was also run to distinguish the contribution of externally and internally generated $PM_{2.5}$ to indoor concentrations. The primary reason to distinguish between these $PM_{2.5}$ sources is the differences in the nature of the particles of indoor and outdoor origin (Adgate et al., 2007; Abdallah et al., 2013) and in their potential (but largely unquantified) relative toxicity, which are sufficiently great as to require separate consideration (Long et al., 2001; Ebelt et al., 2005; Stanek et al., 2011; Rohr and Wyzga, 2012). This gives health impact assessments the opportunity to distinguish relative risks to population health. Emission inventories are based on data from Ozkaynak et al. (1996); He et al. (2004) and Afshari et al. (2005).

Table 5.4 $PM_{2.5}$ sources and schedules used for the specification of baseline simulations of the 2010 and 2050 housing stock.

Common to all simulations of current (2010) and future (2050) stock	Dwelling type	House or Flat
	$PM_{2.5}$ sources and schedules	Models run (1) with $PM_{2.5}$ source and (2) no source scenario
	Cooking	15 mins morning and 30 mins evening cooking, with an additional lunch period of 30 minutes at weekends (gives 1.6 $mg.min^{-1}$ emissions of $PM_{2.5}$)
	Smoking	2 cigarettes in the kitchen on weekdays and weekends and 4 cigarettes on weekdays and 7 at weekends in the living room (giving 0.99 $mg.min^{-1}$ emissions of $PM_{2.5}$ at 5 minutes per cigarette)
	Sweeping	Entrance/bathrooms and en-suites on Wednesday and Saturday only 5minutes per room (giving 0.05 $mg.min^{-1}$ emissions/re-suspension of $PM_{2.5}$)
	Vacuuming	All other rooms on Wednesday and Saturdays only, 5 minutes per room in rotation (giving 0.07 $mg.min^{-1}$ emissions/re-suspension of $PM_{2.5}$)
	Dusting	All rooms Saturdays only, 20 minutes per room in rotation (giving 0.09 $mg.min^{-1}$ emissions/re-suspension of $PM_{2.5}$
	Washing Machine	In Kitchen, scheduled for 30 minutes, 3 times a week (giving 0.12 $mg.min^{-1}$ emissions of $PM_{2.5}$)
	Washing/ Showering	Bathroom and En-suite, daily morning and evening schedule for 30 minutes (giving 0.04 $mg.min^{-1}$ emissions of $PM_{2.5}$)
	Weather	CIBSE/Met Office hourly weather data - Test Reference Year and Design Summer Year

For scheduling of domestic activities and in the absence of suitable data, fixed periods and durations for all domestic activities based on normal behaviour were assumed (Table 5.4). In dwellings occupied by a smoker, and using data from the Office of National Statistics data (ONS, 2000), it was assumed 1 cigarette was smoked per waking hour; the schedule assumes smoking occurs both outdoors and indoors in the kitchen and living room.

For outdoor $PM_{2.5}$ concentrations, the mean annual average concentration was calculated using data downloaded from the 20 urban background monitoring stations in the Automatic Urban and Rural Network (AURN) for London and the London Air Quality Network (LAQN, 2010). These stations had over 95% coverage and ratified data. The annual mean $PM_{2.5}$ concentration was 13µg.m^{-3} with a variance of 2.9 µg.m^{-3} (full details of the calculations are shown in Appendix D).

5.1.3 Personal Exposure and Occupancy Schedules

Personal exposure to $PM_{2.5}$ was estimated from the simulations for three categories of occupancy schedule: (a) a 'household average' concentration of $PM_{2.5}$ in the living room, bedroom and kitchen using time weighting factors of 0.45, 0.45, and 0.1 respectively, based on the relative importance of these rooms both in terms of sources of emissions and assumed times spent in these locations (Wilkinson *et al*., 2009); (b) the exposure experienced by a 'cook' who occupies the living room, bedroom and kitchen during periods of cooking using weighting factors of 0.56, 0.36 and 0.08 on weekdays and 0.4, 0.5 and 0.1 at weekends, respectively; and (c) the exposure of a person who never enters the kitchen and only spends time in the living room and bedroom with weighting 0.62 and 0.38 on weekdays and 0.58 and 0.41 at weekends, respectively. Weighting factors for occupancy schedules (b) and (c) were derived by adjusting the time spent in relevant rooms from the central case (a) from Wilkinson et al., 2009. In order to obtain these outputs, a 'macro' was devised in Excel, that investigated the $PM_{2.5}$ exposure outputs in the various rooms and at different times, simulating the movement of an individual around the dwelling. For each category of occupant, the mean indoor $PM_{2.5}$ exposure across the London stock was calculated from the simulations using weightings that reflect the frequency (proportion) of the permeability distribution within the UK domestic stock (Stephen, 1998) and also that 50% of current (2010) London dwellings are flats, and 50% are houses (ONS, 2011).

5.1.4 Ventilation Modelling

All model simulations assume that dwelling permeability is provided by adventitious openings (gaps and cracks) in the external walls, floors and roofs, with gap size proportional to facade area and assuming a crack is situated at the base and top of each wall (Orme and Leksmono, 2002). A penetration factor (*P*) of 1 was used for all infiltration pathways representing the maximum for $PM_{2.5}$ as CONTAM does not allow the setting of more than one rate for the different types of opening. However, as component size, indoor/outdoor pressure differences, penetration geometry and roughness may affect this factor; sensitivity analysis (section 5.1.7) contrasts this value with a minimum value of 0.6 from Chen and Zhao (2011).

Extractor fans, trickle-ventilators and MVHR systems (where present) were specified to comply with Approved Document F of the Building Regulations for England (HM Government, 2010b). This document stipulates boost flow rates in kitchens and bathrooms during such events as cooking and washing and background rates at other times to achieve minimum whole building ventilation. All equipment was assumed to be correctly fitted, used and perfectly functioning with no deterioration allowed for over time. It is acknowledged that interventions may not be carried out properly, or function to their correct capacity either at the point of construction/commissioning or may deteriorate with time if no properly maintained or if of inferior quality.

For the *present day* (2010), stock models were constructed with two alternative ventilation strategies:
(1) Ventilation achieved via adventitious openings, trickle ventilators, intermittent extract fans and MVHR in line with current building regulations, and periodic purge ventilation by window opening (representing 20% of stock, which has been refurbished, or constructed in line with current regulations).
(2) Ventilation achieved via adventitious openings and periodic purge ventilation by window opening but without trickle ventilators or extraction fans (80% of stock).
These proportions of the stock were informed by the number of dwellings built post 1990 when amendments to Part F of the Building Regulations (1990) were introduced requiring trickle ventilation and extract fans, and data from the Warm Front study, which estimated the percentage of pre-1990 properties already fitted with trickle vents and assumed to have intermittent extract fans (ONS, 2011; Warm Front, 2011).
Outdoor and indoor conditions are a key factor influencing window opening behaviour, however such behaviour is subject to high uncertainty (Andersen et al., 2009). In the absence of specific data, a reasonable assumption was made: for the London stock, a seasonal variation was assumed where windows were opened to 10% of the maximum aperture for 8 hours in the daytime during the summer months and closed during the winter months, except during purge events for cooking, showering and bathroom and toilet usage. Sensitivity analysis was also run on this base case (se section 5.2.3). Consequently, for every scenario separate winter and summer files were produced, which were run in conjunction with the weather files for winter and summer months. Results for $PM_{2.5}$ concentrations were combined in post-processing to calculate yearly average $PM_{2.5}$ concentrations.
A variable wind pressure method (dependent on wind speed and direction) was applied to all building openings in the CONTAM modelling. A wind pressure profile (Figure 5.4) was applied to account for wind direction relative to walls and openings. 12 angular /pressure coefficient pairs were used with the curve fit 2 option in CONTAM connecting all data points using a non-linear (cubic spline) fit between the points (Swami and Chandra, 1987).

Dynamic transient yearly weather files for CONTAM were derived from the CIBSE/Met Office Test Reference Year (TRY) and Design Summer Year (DSY) weather files and were used for the model runs for current day conditions in order to contrast different annual weather patterns. The TRY file is a synthesized typical weather year suitable for analysing the environmental performance of buildings, whereas the DSY is a complete historical year representing a near extreme warm summer (CIBSE, 2010). These files contain hourly outdoor air temperature, air pressure, wind speed, wind direction, and humidity data. Wind profiles within the TRY and DSY files are based on readings at a fixed height (10 metres) in a non-urban situation, usually airports. In order to adjust these to be appropriate for the urban environment, wind speed modifiers were applied based on building height and adjusted for an urban location with flats assumed to be on the ground floor for the purpose of annual average pollutant modelling. Changes in external $PM_{2.5}$ with height (vertical stratification) is investigated separately in section 5.1.6. Transient indoor temperature profiles used for indoor temperatures were informed by a study from FMNectar (2007), which investigated ventilation effectiveness in support of Part F of the Building Regulations. They have a range of average temperatures between zones of (18.75-20.35°C) for winter and (23.65-24.35°C) for summer scenarios.

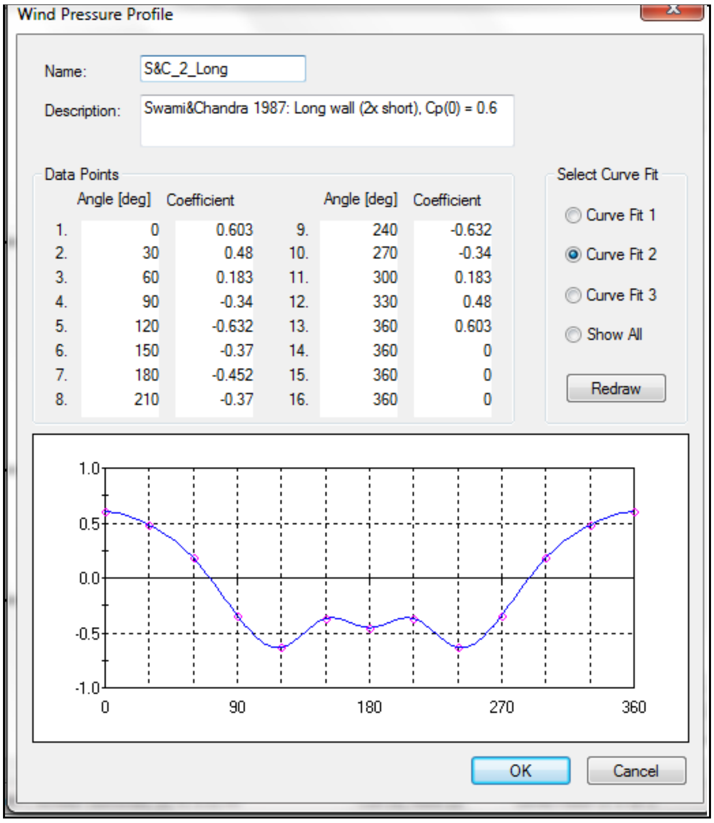


Figure 5.4 Wind pressure profile used in CONTAM modelling

As shown in table 5.4, eight levels of permeability were used for exterior façades: 3, 5, 7, 10, 15, 20, 25 and 30 $m^3m^{-2}hr^{-1}$ at 50Pa. These values reflect the observed distribution of the UK domestic stock (Stephen, 1998, 2000) and are in agreement with later studies showing little change occurring in new build properties (Grigg, 2004). As such, this distribution of permeabilities are assumed also to be broadly representative of the London stock. Internal walls are considered impermeable. Doors when shut, have small gaps modelled between door and frame. Internal doors (excluding storage) are always open except during activities such as cooking and bathroom use. For 'new' dwellings, a gap beneath the door exists to allow air flow for correct functioning of the MVHR system in line with Approved Document F (2010).

For the 2050 housing stock, it is assumed that all dwellings were refurbished and made airtight to a permeability of $3m^3m^{-2}hr^{-1}$at 50Pa and were ventilated by MVHR systems with filters that remove 80% of $PM_{2.5}$ assuming a small degree of inefficiency. The choice of this scenario was motivated by UK targets which aim to reduce CO_2 emission by 80% by the year 2050 (CCC, 2011). The assumption of a complete building refurbishment to this standard and installation of MVHR to all of the London stock is a deliberately extreme scenario to illustrate the maximum feasible impact on $PM_{2.5}$ concentrations in the indoor environment, whilst acknowledging a range of intervention measures are likely. For outdoor $PM_{2.5}$ a concentration of 9µg.m^{-3} was assumed in 2050 based on the impact of a series of emissions policies as noted by Williams (2007). To investigate the effects of a changing climate on the future (2050) scenario, weather files were created by adjusting current day TRY and DSY files in line with climate models based on particular emission scenarios from UK Energy Research Centre (UKERC, 2009) using the method proposed by Belcher *et al.* (2005). However, as these showed no significant impact (< 0.1%) on annual average indoor $PM_{2.5}$ concentrations based on the energy efficiency scenarios used, weather files representing 2010 were used throughout this study.

5.1.5 Model Optimisation to Improve Model Efficiency

Detailed stock modelling requires multiple models to include all the various options and uses considerable computer processing time. Consequently, the purpose of CONTAM model optimisation is to ensure outputs are precise, but achieved at the minimum run time possible. CONTAM allows adjustment of the simulation parameters including calculation time, outputs and status. A low calculation time is critical so as not to miss pollution episodes within zones, however the time can be increased slightly without any detrimental impact on the results. Pre-testing of simulations with various options showed that the combination of a 10 second calculation time, a 15-minute output with a status of 1-hour computation time for a year (summer and winter files combined) gave sufficiently precise (less than 0.01% difference when compared to a 1 second calculation time) averaged $PM_{2.5}$ profiles for each zone in all geometries. Examples of pre-testing are shown in Appendix E.

In order to speed-up the computation time and to avoid possible input errors, models were batch run using an Excel macro ‘CONTAM-batch’ (see Appendix F for details) developed for this process and run using Strawberry Perl, 2008. Perl is a programming language suitable for writing simple scripts as well as complex applications. Strawberry Perl is a Perl environment for MS Windows enabling the development and running of Perl applications (Strawberry-Perl, 2008).

5.1.6 Impacts of Location Relative to Pollutant Source

Additional investigations of the effect of location on external $PM_{2.5}$ concentrations, that could flow into buildings within the Greater London authority (GLA) were carried out using the Operational Street Pollution Model (OSPM, 2010: Vardoulakis et al., 2007). Dwellings were classified into three broad exposure categories – high, moderate and low – based on distance from busy streets and/or intersections (Vardoulakis *et al*. 2008). OSPM was run using composite meteorological and urban background $PM_{2.5}$ files of 13µg.m^{-3} for the present day, based on LAQN data (LAQN, 2010) and 9µg.m^{-3} for 2050, based on estimates from Williams (2007). Research on the LAQN and AURN identified Hackney Clapton Urban Background (UB) Monitoring Station as having the nearest yearly average to this figure, (13.1 µg.m^{-3}) and also approximates to the mean of the UB stations in London that have full data sets for 2010. Additionally, the data from this station has been fully ratified and is therefore more reliable. A dynamic file was composed by downloading the raw hourly raw data from the LAQN as a csv file and pre-processed into a usable format. Negative values seen in the data were removed. Of a total of 8761 possible readings, 22 fell into this category, representing > 0.003%. In order to produce a complete file, gaps in the data were filled by considering figures either side of the gap and comparing them with full days' sets where similar values occurred in hourly profiles and these numbers were substituted. Of a total of 8761 readings, 327 fell into this category, representing > 0.04% which was deemed an acceptable level of possible error due miniscule impact on results. A factor was added to reduce the yearly $PM_{2.5}$ concentrations from 13.1 to 13.0 µgm^{-3} and to produce the future (2050) file at 9 µgm-3. Vehicle traffic and emission data for major non-motorway roads (referred to as A Roads) and minor roads for the same years were constructed with typical London street configurations (Vardoulakis et al., 2008).

Residences in London were subdivided into houses and flats and further classified in three broad exposure categories (very high, high, moderate and low) based on distance from busy streets and/or intersections as explained by Vardoulakis et al. (2008). OSPM was run using composite meteorological files for the years 2010 and 2050, vehicle traffic and emission data for major ("A") and minor roads for the same years, and typical London street configurations ("A" road or "minor" road flanked by houses or blocks of flats, with the prevailing wind parallel or perpendicular to the street axis).

Traffic data for "A" roads and minor roads in London were based on the UK Department for Transport's Road Traffic Statistics (2005). All data were projected to 2010 and 2020 using a linear extrapolation of 1993 to 2005 trends. Data were then assumed to remain constant from 2020 to 2050 due to the large uncertainties after this date and assuming no further change in vehicle-kilometres driven beyond 2020 (Williams, 2007). Vehicle fleet composition information for the UK was scaled to represent urban "A" and minor roads using a constant adjustment based on differences between 2005 UK and urban data (London-specific fleet composition data were unavailable). Similarly, vehicle flows for the two road types were scaled to represent London, based on differences between national and London flows for 2005. Average weekday peak and inter-peak traffic speeds for 2009 were taken from Transport for London's latest strategic speed survey, covering the years 2007 to 2009 (TfL, 2010). Non-peak traffic speeds were assumed to be 30 mph (48.3 km/h). Traffic speeds in 2020 (and 2050) were assumed to have decreased by 4%, a figure broadly consistent with changes observed by Transport for London

(TfL, 2010). The traffic information was used to estimate $PM_{2.5}$ emissions using the Emission Factor Toolkit (EFT, 2010) version 4.2, which calculates emission rates for given vehicle and traffic flow information. The Toolkit is based on emission factors published by the Department for Transport and incorporates emissions from vehicle exhausts and brake/tyre wear.

OSPM output data for two receptors, one on each side of the street, were averaged so as to obtain a single $PM_{2.5}$ value for each typical London street significantly affected by road traffic. An intersection "enhancement" (corresponding to a 25% increase of the modelled roadside $PM_{2.5}$ contribution) was added to the calculated outdoor concentration for buildings within 100 m from busy intersections, while residences at distances greater than 150 m away from busy roads were assigned to the urban background levels (Vardoulakis et al., 2008). All outdoor $PM_{2.5}$ concentrations were simulated at a receptor height of 1.5 m above the ground for houses, and at receptor heights of 1.5, 3, 6, 9 and 13.5 m for typical blocks of flats in London. The final concentrations obtained were checked for consistency against annual mean $PM_{2.5}$ concentrations observed at different roadside air quality monitoring sites in London in 2010 and previous years (Charron and Harrison, 2005; LAQN, 2010). Key model inputs are summarized in Table 5.5a vehicle traffic and 5.5b street characteristics.

Table 5.5a OSPM inputs: vehicle traffic

Vehicle traffic	"A" Road		Minor Road	
	2010	2050	2010	2050
AADT*	29,190	30,072	2,782	3,113
Average HGV%	6.69%	4.96%	4.06%	3.25%
Average speed	39.58 km/h	38.03 km/h	39.58 km/h	38.03 km/h
$PM_{2.5}$ emission rate	34.38 g/km	19.79 g/km	3.07 g/km	1.98 g/km

* Annual average daily traffic

Table 5.5b OSPM inputs: street characteristics

Street characteristics	"A" Road		Minor Road	
	House	Flat	House	Flat
Street width	34 m	34 m	20 m	20 m
Building height	12 m	24 m	12 m	24 m
Street length	300 m	300 m	200 m	200 m
Receptor height	1.5 m	1.5, 3, 6, 9 and 13.5m	1.5 m	1.5, 3, 6, 9 and 13.5m
Street orientation	parallel: 30 degrees & perpendicular: 120 degrees			

5.1.7 Sensitivity Analysis of Estimates of $PM_{2.5}$ Concentrations

The current levels of personal $PM_{2.5}$ exposure and those following the application of energy efficiency interventions and ventilation strategies are not currently known with any degree of certainty. This thesis uses the best published empirical data for modelling inputs such as emission rates and considers relative rather than absolute changes in concentrations of $PM_{2.5}$. The investigation of model uncertainties via sensitivity analysis on model inputs yields a distribution of $PM_{2.5}$ concentrations and therefore potential exposures, compared to single values for individual interventions. When predicting indoor $PM_{2.5}$ exposures using computer simulation, uncertainty in the predicted exposures needs to be taken into account. Sensitivity analysis is used to calculate uncertainties in the programme outputs which are caused by those in both the computational processes and in data used as input variables (Dutton, et al., 2008).

Differential sensitivity analysis (DSA) was carried out to examine the sensitivity of the results to model inputs and assumptions, and specifically to locate those variables which were the most influential. This method assumes that the effect of each variable is independent and additive. For numerical parameters (e.g. $PM_{2.5}$ emission and deposition rates) high and low values were calculated as the means $\pm$2.33 standard deviations - the range that encompasses 99% of the values assuming normally distributed data (Lomas and Eppel, 1992). For other variables such as window opening, where field data are sparse, proposed values sought to reflect the range of normal behaviour, changing the central values for open period by $\pm$2 hours and increasing the aperture to 40% of the total window area. For indoor temperature schedules were adapted $\pm$2°C. although it is acknowledged that this does not comply with the principle of 'means $\pm$2.33 standard deviations', it was felt that in the absence of clear data, these values represented as near as possible the 95% confidence level, although further research is needed to confirm/refute this assumption. For building orientation, the dwellings were rotated in steps of 45 degrees. Model runs using different weather files (Heathrow TRY, Heathrow DSY and Gatwick International Weather for Energy Calculation (IWEC) (ASHRAE, 2010), were performed to consider possible effects of locational meteorological variation (CIBSE, 2010). Other analyses examined the effect of altering the height of the infiltration gaps by $\pm$0.1m from the initial height of 2.3m. Variation in room volumes was informed by Chapman (1994) based on the range of storey heights (2.3-2.6m) within the GLA. These yield average room volume changes of $\pm$8.6% from the baseline models. In the case of building orientation, the mean deviation from the base line value (north) was calculated. Full details are shown in Table 5.6. This entailed the construction/alteration of the 135 baseline Contam models and their variants to produce a further 270 independent models to investigate sensitivity analysis.

Table 5.6 Variables adjusted during sensitivity analysis, their changes from standard values and source.

Variable	Standard value	Change in value	Source
Room volume	Height 2.4m	+/- 0.1m +/- 4.3%	Chapman, 1994
Crack spacing	2.3m	+/- 0.1m +/- 4.3%	Assumed
Building Orientation	N-S	Rotate in 45° intervals	Assumed
Indoor Temperature	Daily Schedule	Alter by +/- 2°C	Assumed
Internal $PM_{2.5}$ source	1.6 mg/min	+/- 0.6, 1.2; 1-2.33 std deviations	Dimitroulopoulou et al., 2006ed
Deposition Velocity	0.00018 m/s	+/-0.00010 1-2.33 std deviation	Dimitroulopoulou et al., 2006
External $PM_{2.5}$	9 &13µgm/m^3	9-21 µgm/m^3	LAQN, 2011
Permeability	3-30 m^2/hr@50Pa	Apply all values within the UK distribution	Stephen, 1998

The maximum and minimum alternative values of each input parameter (*p*) were entered into the model while holding all other variables constant. The results are reported as the percentage difference in $PM_{2.5}$ concentrations compared with the central baseline estimate for each variable. The quadrate sum (Δptot) is calculated from the square of the values obtained to give the total variation from the base value as seem in equation 8:

$$\Delta p\text{tot} = \left(\sum_{t-1}^{1} \Delta pi^2\right) \qquad \text{Equation 8}$$

Where

Δptot is the predicted parameter

$(\sum_{t-1}^{1} \Delta pi^2)$ is the quadrate sum of individual variables being investigated.

Assuming that the uncertainty in the predicted parameter is normally distributed, the value of Δptot is only strictly correct if the sensitivity to each individual input is independent of the value of the other inputs i.e. the computer program behaves as a superposable system. For most systems this is not strictly true. Nevertheless, for small changes in the input data the assumption may be considered reasonable (Lomas and Eppel, 1992). Further aspects of uncertainty and assumptions within the modelling are covered in Appendix E.

5.1.8 Health Impact Assessment

Assessment of the potential impacts on health of changes in indoor exposure to $PM_{2.5}$ was performed by Dr James Milner at the London School of Hygiene and Tropical Medicine (LSHTM) using Excel incorporating population data for the Greater London Area (GLA), smoking rates and $PM_{2.5}$ outputs from the CONTAM modelling for various occupants combined with the life table model IOMLIFET (Miller and Hurley, 2006). The description of the work carried out by staff at LSHTM using GLA and

$PM_{2.5}$ data supplied by the author is given here in order to clarify the method used to calculate health impacts. This method estimates changes in population survival (mortality and not morbidity) as a result of changes in risk associated with environmental exposures. For the modelled scenarios, impacts of changing $PM_{2.5}$ exposures on cardiopulmonary and lung cancer mortality were conducted. Age-specific population data and rates for all-cause and disease-specific mortality for England and Wales for the present (2010) scenario were obtained from the UK Office for National Statistics (ONS, 2010). The life table outputs were subsequently scaled for London using population data from the Greater London Authority (GLA, 2010).

To determine the specific impact on cardiopulmonary and lung cancer mortality, the corresponding $PM_{2.5}$ mortality coefficients from the American Cancer Society (ACS) cohort study of air pollution and health were used (Pope et al., 2002). These show an 8.2% increase in cardiopulmonary mortality per 10 $\mu g.m^{-3}$ increase in annual average $PM_{2.5}$ and a 5.9% increase for lung cancer mortality. For consistency with the age profile of participants in the ACS study, the increased risk has been applied to adults aged 30 and over, although there is evidence from elsewhere of $PM_{2.5}$ effects on mortality in children (e.g. Woodruff et al., 2006). It is therefore entirely possible that this method underestimates the total likely overall risk. The health impact calculations have been conducted on the total indoor exposure to $PM_{2.5}$ based on indoor exposure to $PM_{2.5}$ derived from indoor and outdoor sources. However, it is acknowledged that there is some debate over their relative risks to population health where ongoing (but largely unquantified) uncertainties in the relative toxicities exist (Long et al., 2001; Ebelt et al., 2005; Stanek et al., 2011; Rohr and Wyzga, 2012). Mortality rates and population size for 2010 were used for the 2050 scenario calculations due to uncertainties in future population characteristics. Previous studies have shown impacts modelled using life tables are relatively insensitive to baseline population rates (e.g. Miller and Hurley, 2006). Health burden calculations follow the method seen in (COMEAP, 2010). The primary outputs of the analysis are changes in attributable deaths for the London population over their lifetime and changes in life expectancy at birth.

5.2 Results

5.2.1. Building Stock Modelling

The results shown below represent the mean annual average indoor $PM_{2.5}$ exposures for London for the population as a whole in the present day (2010) and the changes that occur in a possible future scenario once MVHR and reductions in permeability are achieved (2050). The different exposure models represent those experienced due to different occupant activity patterns, with the household average shown for reference. Rather than consider peak $PM_{2.5}$ events, this approach is taken in order to investigate possible long-term health impacts which require annual $PM_{2.5}$ exposure figures.

5.2.1.1 Non-Smoking Households

The results of the CONTAM simulations of $PM_{2.5}$ exposure for non-smoking households are presented in Table 5.7.

Table 5.7 London dwellings without smoking occupants: simulated and weighted average annual indoor $PM_{2.5}$ exposures for the present day (2010) and 2050.

Year	Exposure Model	Indoor exposure to $PM_{2.5}$ from indoor sources $\mu g.m^{-3}$ (percentage of total)	Indoor exposure to $PM_{2.5}$ from outdoor air $\mu g.m^{-3}$ (percentage of total)	Total $\mu g.m^{-3}$	Change in total indoor $PM_{2.5}$ 2010-2050 $\mu g.m^{-3}$
Present Day (External $PM_{2.5}$ 13.0 $\mu g.m^{-3}$)	Household Average	22.0 (77%)	6.4 (23%)	28.4	
	Cook	54.2 (90%)	6.3 (10%)	60.5	
	Non-cook	9.4 (61%)	6.1 (39%)	15.5	
2050 (External $PM_{2.5}$ 9.0 $\mu g.m^{-3}$)	Household Average	8.2 (85%)	1.4 (15%)	9.6	-66%
	Cook	16.5 (92%)	1.4 (8%)	17.9	-70%
	Non-cook	3.1 (70%)	1.3 (30%)	4.4	-71%
2050 Permeability change only	Household Average	27.8 (82%)	5.0 (18%)	33.8	+19%
	Cook	65.4 (93%)	4.9 (7%)	70.3	+16%
	Non-cook	13.1 (73%)	4.8 (17%)	17.9	+15%

The simulations suggest that under present day conditions, average indoor concentrations of $PM_{2.5}$ are appreciably higher than those in the outdoor air because of indoor sources. Thus, in non-smoking dwellings, although indoor levels of $PM_{2.5}$ derived from outdoor air are less than half the outdoor levels, the concentration experienced by the average household member indoors was estimated to be 28.4 $\mu g.m^{-3}$, over twice the concentration in the outdoor air (13.0 $\mu g.m^{-3}$). Most of the contribution to this very high level of indoor particle exposure was from cooking-related sources as indicated by the difference in exposure of the cooks and non-cooking occupants. A modelling scenario with reductions in envelope permeability without MVHR produced further increases in indoor $PM_{2.5}$ concentrations; 5.4µg.m-3 for typical household members and 9.8µg.m-3 for cooks in both smoking and non-smoking households.

Under the 2050 refurbishment scenario, household average exposure to total $PM_{2.5}$ (from indoor and outdoor sources) was reduced from 28.4$\mu g.m^{-3}$ to 9.6$\mu g.m^{-3}$ (-66%) as a result of permeability reductions, the application of correctly installed and perfectly functioning MVHR equipment and a reduction in outdoor concentrations in the 2050 scenario. The contribution from external sources represents 23% of the current total indoor $PM_{2.5}$ exposure and 15% in 2050. Average London domestic stock indoor/outdoor (I/O) ratios for $PM_{2.5}$ from external sources are 0.5 for present day and 0.2 for 2050 due to the decrease in stock permeability and filters on the MVHR system. Separate 2050 scenarios with the proposed reduction in permeability to $3m^3m^{-2}hr^{-1}$ at 50 Pa but *without* providing an MVHR system, results in an increase in the London annual average indoor exposure to total $PM_{2.5}$ of 5.4μg.m^{-3} from the baseline of 28.4μg.m^{-3}. For cooks, under the same scenario, the increase is 9.8μg.m^{-3} from the baseline of 60.5μg.m^{-3} and for non-cooks, an increase of 1.5μg.m^{-3} from 15.5μg.m^{-3}. These increases are due to the influence of decreases in outdoor $PM_{2.5}$ penetration and reduced ventilation of the $PM_{2.5}$ from indoor sources.

There was considerable variation in $PM_{2.5}$ exposure levels among household members. The simulations show that in non-smoking households peak exposure levels are related to periods of cooking in the kitchen as noted by others (Ozkaynak, et al., 1996; Weschler, 2009). The results suggest cooks may experience twice the level of $PM_{2.5}$ exposure of the average household member, and more than four times that of a 'non cook' who does not enter the kitchen. This is because the average cook is exposed to 5.8 times the internally generated $PM_{2.5}$ compared with the average non cook, while both are exposed to roughly similar levels of externally generated $PM_{2.5}$. The household average $PM_{2.5}$ exposure (the time-weighted average of $PM_{2.5}$ experienced from occupancy in the living room 45%, bedroom 45% and kitchen 10%) approximates the average exposure of a family of one cook and three non-cook members (average exposure=26.8μg.m^{-3}).

5.2.1.2 Smoking Households

The results of the CONTAM simulations of $PM_{2.5}$ exposure for smoking households are presented in Table 5.8.

Table 5.8 London dwellings with smoker occupants: simulated average annual indoor $PM_{2.5}$ exposures for the present day and 2050

Year	Exposure Model	Indoor exposure to $PM_{2.5}$ from indoor sources μg.m^{-3} (percentage of total)	Indoor exposure to $PM_{2.5}$ from outdoor air μg.m^{-3} (percentage of total)	Total μg.m^{-3}	Change in total indoor $PM_{2.5}$ 2010-2050 μg.m^{-3}
Present Day (External $PM_{2.5}$ 13.0 μg.m^{-3})	Household Average	51.4 (89%)	6.4 (11%)	57.8	
	Cook	96.0 (94%)	6.3 (6%)	102.3	
	Non-cook	51.0 (95%)	6.1 (5%)	57.1	
2050 (External $PM_{2.5}$ 9.0 μg.m^{-3})	Household Average	26.5 (95%)	1.4 (5%)	27.9	-52%
	Cook	46.1 (97%)	1.4 (3%)	47.5	-54%
	Non-cook	32.8 (96%)	1.3 (4%)	34.1	- 40%
2050 Permeability change only	Household Average	63.8 (93%)	5.0 (%)	68.8	+19%
	Cook	113.8 (96%)	4.9 (%)	118.7	+16%
	Non-cook	60.9 (93%)	4.8 (%)	65.7	+15%

According to the English Housing Survey 2009, the proportion of properties in London with smokers is 19% (EHS, 2009). For smoking households, the concentration of $PM_{2.5}$ experienced by the average household member was 57.8µg.m^{-3}, over four times the outdoor concentration. The external $PM_{2.5}$ component now represents a substantially smaller proportion (11%) of the overall exposure compared to the non-smoking scenario. The non-cook receives a similar exposure to the average household member (57.1µg.m^{-3}) due to $PM_{2.5}$ emissions from smoking occurring in the living room. The cook experiences an annual average increase in $PM_{2.5}$ exposure of +41.8µg.m^{-3} compared to the non-smoking scenario (60.5 to 102.3µg.m^{-3}). The 2050 refurbishments reduce the indoor exposure substantially. However, with reduced permeability and the MVHR system designed according to Approved Document F (HM Government, 2010b) occupants still experience exposures between 3.1 and 5.3 times the external $PM_{2.5}$ concentration of 9.0µg.m^{-3}.

5.2.2 Location Analysis

The results of the OSPM modelling are shown in Table 5.9.

Table 5.9 External $PM_{2.5}$ concentrations based on building type, location in relation to traffic sources and receptor height.

Location		Annual Average $PM_{2.5}$ (ug.m^{-3})			
		"A" Road		Minor Road	
Type	Receptor	2010	2050	2010	2050
(Very High) Intersection					
House	1.5m	14.86	10.38	13.09	9.32
Flat	1.5m	14.98	10.46	13.16	9.37
Flat	3.0m	14.83	10.38	13.13	9.35
Flat	6.0m	14.07	9.92	12.96	9.23
Flat	9.0m	13.68	9.68	12.91	9.19
Flat	13.5m	13.37	9.49	12.87	9.17
(High) Street Canyon					
House	1.5m	14.44	10.12	13.02	9.27
Flat	1.5m	14.54	10.19	13.08	9.31
Flat	3.0m	14.41	10.12	13.06	9.30
Flat	6.0m	13.81	9.75	12.92	9.21
Flat	9.0m	13.49	9.56	12.88	9.17
Flat	13.5m	13.25	9.41	12.85	9.15
(Moderate) Background					
House	1.5m	12.76	9.09	12.76	9.09
Flat	1.5m	12.76	9.09	12.76	9.09
Flat	3.0m	12.76	9.09	12.76	9.09
Flat	6.0m	12.76	9.09	12.76	9.09
Flat	9.0m	12.76	9.09	12.76	9.09
Flat	13.5m	12.76	9.09	12.76	9.09

The worst case scenario results show a maximum $PM_{2.5}$ external variation of +1.98μg.m^{-3} against an urban background value of 13.0μg.m^{-3} (flats at major road intersections, with a receptor height of 1.5m). In the future 2050 setting, this same scenario shows a variation of +1.46μg.m^{-3} against an urban background value of 9.0μg.m^{-3}. Considering the average exposure in a non-smoking household, this represents an increase in total indoor exposure of 0.9 μg.m^{-3} compared to central values for the 2010 stock and an increase in total indoor exposure of 0.2 μg.m^{-3} compared to central values for the 2050 stock.

5.2.3 Sensitivity Analysis

The results of the sensitivity analysis are shown in Table 5.10. Figures indicate the percentage changes in average annual indoor personal $PM_{2.5}$ concentrations when each of the listed parameters took high or low values with all other input parameters held constant. All models studied were for present day London stock (2010) based on the 45%, 45%, 10% household occupation scenario. The percentage change in $PM_{2.5}$ concentrations for each variable represents its independent effect on the baseline model results. The results from the location modelling have been added to those from the CONTAM modelling in the following table. The quadrate sum is calculated from the square of the values obtained, enabling the overall error in the baseline values to be calculation using the method from Lomas and Eppel (1992).

Table 5.10 Effect of assuming high and low values for key input parameters in model simulations.

Variable	Average annual GLA % difference in $PM_{2.5}$ estimate
$PM_{2.5}$ Deposition Rate	$\pm$59.1
$PM_{2.5}$ Emission Rate	$\pm$36.2
Window Opening	$\pm$18.3
$PM_{2.5}$ Infiltration Rate	$\pm$9.0
Volume	$\pm$4.2
Building Orientation	$\pm$3.7
Location Analysis OSPM	$\pm$3.2
Weather File Changes	$\pm$1.3
Indoor Temperature	$\pm$0.6
Infiltration Height	$\pm$0.2
Quadrate Sum	*$\pm$72.5*[a]

[a]In smoking houses this rises to $\pm$ 106.8% from the base line rate based on the increase in $PM_{2.5}$ emissions from smoking.

Indoor $PM_{2.5}$ deposition and emission rate and window opening behaviour had the largest influence on the overall $PM_{2.5}$ concentrations, with generally smaller impacts from building orientation, infiltration height, volume, indoor temperature, external weather conditions and building location within the GLA. The quadrate sum for smoking ($\pm$106.8%) and non-smoking properties ($\pm$ 72.5%) showed very large variations in $PM_{2.5}$ exposure are possible, although in any given situation some of these factors may cancel each other out. It should be stressed that there is also variation in the uncertainty of the individual parameters used. For example, variability in building height and volume (Chapman, 1994) is better understood than variability in behaviour and window opening (Andersen et al., 2009), which are driven by the assumptions within the models.

5.2.4 Health Impact Assessments

The results of the health impact assessments reflecting the exposure model used for 2010 and 2050 are shown in Tables 5.11 and 5.12.

Table 5.11: Modelled health impacts in London in non-smoking homes (81% of homes)

Year	Exposure Model	Total annual attributable deaths	Life expectancy at birth: years lost (days)		Change in Total annual attributable deaths
			Male	Female	
Current Year 2010	Household Average	6,877	1.32 (484)	1.25 (457)	
	Cook	5,207	2.81 (1,024)	2.64 (964)	
	Non-cook	2,385	0.73 (265)	0.69 (251)	
2050	Household Average	2,465	0.45 (164)	0.43 (156)	-64%
	Cook	1,737	0.84 (306)	0.79 (289)	-67%
	Non-cook	699	0.21 (75)	0.20 (72)	-66%

*Household average is based on occupancy as follows; 45% bedroom, 45% living room, 10% kitchen

As is to be expected, the levels of exposures seen in Table 5.7 are reflected in the total attributable deaths seen with the greatest impact on cooks in the 2010 scenario. Substantial health gain are possible in the scenarios investigated.

Table 5.12: Modelled health impacts in London in non-smoking homes (19% of homes)

Year	Exposure Model	Total annual attributable deaths	Life expectancy at birth: years lost (days)		Change in Total annual attributable deaths
			Male	Female	
Current Year 2010	Household Average	3,009	2.68 (979)	2.53 (922)	
	Cook	1,835	4.73 (1,725)	4.44 (1,620)	
	Non-cook	1,822	2.65 (967)	2.50 (911)	
2050	Household Average	1,597	1.30 (475)	1.23 (449)	-48%
	Cook	988	2.21 (806)	2.08 (760)	-65%
	Non-cook	1,160	1.59 (580)	1.50 (548)	-37%

In smoking houses, smoking represents an additional source of $PM_{2.5}$ which also impact non-smokers in the household. Approved Document F, specifically states that it has *not* been formulated to deal with the products of tobacco smoking ADF (2010). Even with a ventilation strategy (MVHR) complying with ADF, 2010, and using the relatively conservative rates of indoor smoking in this study, there are still substantial negative health impacts on a small percentage of the population (19%) from smoking.

Due to the smoking occurring in various rooms within the property, the variation in health impacts between cooks and non-cooks is not so pronounced as in non-smoking homes.

5.3 Discussion

This section provides new insights into the potential effect of changes to the energy efficiency of London's housing stock on exposure to $PM_{2.5}$, and possibly even the potential for the reduction of other airborne pollutants, by the use of extraction fans, and in addition (although not an energy efficiency measure in its own right) the use of filtration -80% efficiency in this model- of pollutants through mechanical ventilation systems. The results suggest that domestic energy efficiency interventions of the type and scale needed to meet 2050 climate change abatement objectives could yield substantial net reductions in $PM_{2.5}$ exposure. Modelling of the future scenario for non-smoking dwellings show a reduction in annual average indoor exposure to $PM_{2.5}$ of 18.8μg.m^{-3} (from 28.4 to 9.6μg.m^{-3}) for a typical household member. Also of interest is that a larger reduction of 42.6μg.m-3 (from 60.5 to 17.9μg.m-3) was shown for members exposed primarily to cooking-related particle emissions in the kitchen (cooks). Appreciable reductions in $PM_{2.5}$ exposure are also seen in smoking dwellings, thereby also reducing the health impacts for non-smoker in smoking households. In addition, it is assumed that the current mean outdoor $PM_{2.5}$ concentration of 13μg.m^{-3} decreased to 9μg.m^{-3} by 2050 due to emission control policies being achieved. Using the modelled interventions, failure of this policy will have a smaller impact than might otherwise be the case if increases in airtightness and the use of MVHRs did not occur. This smaller impact is due to the reduction of ingress of external $PM_{2.5}$ to the indoor environment (23-11% for the household average scenario). However, the magnitude and directions of exposure changes depend on the details of the specific mitigation measures, and adverse effects may occur, for example, if air-tightness is achieved without the associated installation and maintenance of correctly functioning compensatory ventilation systems. Reductions in envelope permeability without mechanical ventilation, but with compensatory window opening, produced increases in indoor $PM_{2.5}$ concentrations in non-smoking houses of 5.4μg.m^{-3} for typical household members 9.8μg.m^{-3} for cooks and 2.4 μg.m^{-3} for non-cooks, implying that energy savings are made at the price of population health. Similar increases are seen in smoking homes (see tables 5.8 and 5.9) leading to negative health impacts. The results for both smoking and non-smoking households represent only time spent in the indoor domestic environment. In order to quantify overall personal exposure to $PM_{2.5}$ and consequent health impacts, time spent in other microenvironments (e.g. in transport, at work) and outdoors will need to be taken into account (Wallace et al., 2006). These estimates of changes in $PM_{2.5}$ exposure are sensitive to assumptions about occupant behaviour, ventilation system usage and the distributions of input variables ($\pm$72% for non-smoking and $\pm$107% in smoking residences). However, if realised, they would result in significant health benefits. As previously stated, the method used to calculate possible health impacts may underestimate the total likely overall risk from $PM_{2.5}$ exposure, such that any health gains have their own uncertainties associated with exposure changes.

The use of MVHR systems when used in conjunction with tightening of the building envelope could result in a substantial reduction of indoor sources (including tobacco smoke), and markedly lower exposures from outdoor sources. When PPV is included as Part of the retrofit interventions, for the exposures seen, the simulated changes are likely to be positive for health which may add to the case for pursuing energy efficiency interventions with PPV – if properly implemented and maintained. There is much anecdotal evidence that filters are often not changed regularly by occupants, or not at all, thereby decreasing their efficiency and reducing possible health benefits. However, airtightening without PPV has the opposite impact, increasing the levels of indoor pollution, causing negative health impacts.

The results should be interpreted with caution as they are dependent on the range of assumptions and input parameters specified. In particular, field studies show high variability in $PM_{2.5}$ emission rates from cooking: 2.4$\pm$2.1mg.min^{-1} (He et al., 2004), 36$\pm$98mg.min^{-1} (Olson and Burke, 2006), 1.6$\pm$0.6 mg.min^{-1} (Ozkaynak et al., 2006). Similarly, there are variations in deposition rate calculation methodologies with differing interpretations of surface area (Fogh et al., 1997; Thornburg, et al., 2001), which could lead to differences in absolute $PM_{2.5}$ exposures. The sensitivity analysis indicates that there are large uncertainties in $PM_{2.5}$ emissions and deposition rates which influence exposure. Although these would not generally affect the direction of change (lower $PM_{2.5}$ values in the 2050 scenario), they may affect its magnitude, either positively or negatively. The other major factors affecting personal indoor exposure appear to relate to changes to the building envelope, ventilation systems and occupant activity. Results from the OSPM modelling indicate that the location of properties within London are a less critical factor determining indoor $PM_{2.5}$ exposures, when compared to the impacts of the energy efficiency interventions proposed. However, for those current properties with higher permeabilities, they will receive a greater level of $PM_{2.5}$ from outdoor sources. If they are closer to a high pollutant source (e.g. an intersection on an 'A' road) this external component will further increase. A wider range of interventions and building geometries would be needed to be modelled to investigate the distribution of concentrations.

The specification of the 2050 stock was deliberately based on an extreme scenario, with all dwellings reduced in permeability to 3.0m^{3}m^{-2}hr^{-1} and fitted with MVHR systems, combined with effective 80% particle filtration. However, the UK's 'Retrofit for the Future' programme contains examples of construction refurbishment projects in London employing MVHR systems and achieving substantial reductions in permeability, in some cases to the Passivhaus standard of 0.6m^{3}m^{-2}hr^{-1} (LEB, 2011). So, whilst such permeabilities are achievable, the impacts seen on $PM_{2.5}$ levels are therefore likely to be towards the maximum of what could be achieved. They do, however, suggest possible positive impacts on $PM_{2.5}$ exposures from energy efficiency measures implemented as part of a strategy for meeting abatement targets as specified by the UK Committee on Climate Change (CCC, 2011). To make clear the effect of these housing changes, the models assumed no future changes in behaviour or new technologies, which could further influence indoor air quality, save for the assumption of a lower outdoor $PM_{2.5}$ concentration in 2050. However, one possible behavioural impact following retrofitting that could influences its effectiveness as a climate change mitigation strategy is the 'take-back' or Jevons effect, whereby occupants come to prefer the higher temperatures enjoyed and so the savings in energy are underwritten by an increase in energy use such that the emissions reductions expected may not actually materialise (Gillingham et al. 2014),

5.3.1 Comparison with Empirical Studies

The exposure results for the current day are broadly consistent with on-site measurements of average annual indoor domestic $PM_{2.5}$ concentrations. Hanninen et al. (2004) as part of the EXPOLIS project monitored indoor $PM_{2.5}$ concentrations in non-smoking households in four European cities, (some of which may represent an appropriate comparison to London), showing the following variations: Athens 23 ± 11µg.m^{-3}; Basel 17 ± 8µg.m^{-3}; Prague 25 ±16µg.m^{-3} and Helsinki 25 ± 16µg.m^{-3}. It is acknowledged that there are some differences in construction geometries, techniques and materials between the locations. Our estimated non-smoking household average exposure for London of 28.4µg.m^{-3} is slightly higher than other studies. Wallace et al. (2006) monitoring 36 residences in North Carolina over a year show mean indoor $PM_{2.5}$ concentrations of 25.8µg.m^{-3} with a range of 7.2-66.0µg.m^{-3} for non-smoking households. Substantial variations are seen in empirical studies on smoking concentrations including $PM_{2.5}$. For properties where smoking occurred, a consumption of 7.4 cigarettes per day results in an average indoor $PM_{2.5}$ concentration of 132.7µg.m^{-3} measured over 14 days while 4 cigarettes over 19 days yields 66.0µg.m^{-3} (Wallace et al., 2006). The present study shows an average household $PM_{2.5}$ exposure of 57.8µg.m^{-3} for 7 cigarettes per day modelled over a year. This difference may be due to a number of factors including cigarette type and various building parameters, some of which are not available from the study. Dimitroulopoulou et al. (2005) in a monitoring study on kitchens in 37 new homes in the UK with smokers found 24-hour mean $PM_{2.5}$ concentrations of 113µg.m^{-3} in winter and 134µg.m^{-3} in summer compared to an annual average of 102 µg.m^{-3} for the personal exposure of a cook in this study.

5.3.2 Comparison with Other Modelling Studies

The results for current day exposure profiles are generally consistent with those of other published research. Dimitroulopoulou et al. (2006), using the INDAIR probabilistic model calculated annual indoor mean $PM_{2.5}$ concentrations of 19.78g.m^{-3} (calculated to correspond to the base case (a) occupancy scenario, see section 5.1.3) in households with gas cooking, with a peak value of 318µg.m^{-3} (compared with 442µg.m^{-3} for this present study) and a standard deviation of 78µg.m^{-3} in the kitchen.. McGrath et al. (2014) produced the Indoor Air Pollutant Passive Exposure Model (IAPPEM), building on the work of Dimitroulopoulou et al. (2006), and found higher mean concentrations of $PM_{2.5}$ of 166 ± 11 µg.m^{-3}. Fabian et al. (2012), using CONTAM to model low-income multifamily housing without smoking and with a higher cooking emission rate of 1.56mg.min^{-1} calculated a mean indoor $PM_{2.5}$ concentration of 52.9µg.m^{-3} with a standard deviation of 41.4µg.m^{-3}. Emmerich and Howard-Reed, (2005), using CONTAM modelling as part of the U.S Dept. of Housing and Urban Development's Healthy Homes Initiative, found the two most effective intervention strategies for indoor air quality were extract fans, if operated during source events (kitchen fan airflow rate 47 l/s) and efficient air filtration on heating ventilation and air conditioning (HVAC) systems, if operated for a minimum 15% of the time. Our result for the household average $PM_{2.5}$, I/O ratio for the 2050 scenario of 0.15 for an external concentration of 9µg.m^{-3} using an MVHR system at 80% filter efficiency are consistent with results from Mackintosh et al. (2010) with a $PM_{2.5}$, I/O ratio 0.1 for an external concentration of 15µg.m^{-3}.

Future work with a larger set of geometries that are more representative of the London stock could consider a probabilistic approach such as Monte Carlo Analysis (MCA) which was beyond the scope of this study. Unlike DSA, MCA is unable to detect effects of component variables (Dutton, et al., 2008). However, in reality many of the input variables within the CONTAM models are linked, e.g. window opening and air change rates, and as such they are not truly linear and superposable (an assumption of DSA techniques used). MCA could provide an indication of homes with overall characteristics likely to lead to higher indoor concentrations of $PM_{2.5}$ (Dimitroulopoulou et al., 2006).

5.4 Conclusions

This chapter has developed and applied a series of model simulations in CONTAM to quantify the changes in indoor domestic exposure to $PM_{2.5}$ in the Greater London Authority (GLA) Area as a result of the application of specific energy efficiency measures designed to meet 2050 greenhouse gas abatement targets. It has quantified the key variables influencing indoor $PM_{2.5}$ exposure and shown that construction and occupant factors are major influences, with building location within London having by comparison a relatively smaller effect. This study has shown that the CONTAM software and the models produced are suitable for investigating Domestic $PM_{2.5}$ exposure when compared to other empirical and modelling studies, although as previous stated, uncertainties exist based primarily on model assumptions. The methods used in this paper could be applied to assess a wider variety of refurbishment strategies, differing pollutants and occupant schedules in order to analyse the impacts on a wider range of dwelling geometries.

Although there are uncertainties associated with the results, the primary findings suggest possible substantial reductions in $PM_{2.5}$ exposure, which are likely to be beneficial for health in most cases if the interventions are implemented appropriately and include PPV with airtightening. However, without PPV and/or imperfect implementation increases in negative health impacts are likely to occur.
The work confirms that present day high exposures for cooks from particle emissions during cooking in domestic environments are avoidable through a comparatively simple adaptation such as the introduction and use of extraction equipment as noted by Fabian et al. (2012); or by properly fitted, maintained and operated MVHR systems. These also help to remove some of the indoor $PM_{2.5}$ derived from tobacco smoke, which would be beneficial for non-smokers in such dwellings, although this is specifically stated as not the purpose of ventilations standards in the building regulations (ADF, 2010).

This chapter has quantified current and a possible future $PM_{2.5}$ indoor exposure in the London domestic stock and following particular energy efficiency interventions (MVHR and air-tightening) using modelling techniques. There is a need for a broader range of interventions to be explored (e.g. cavity insulation, double glazing, extract fans, trickles vents etc,) with a greater range of representative building geometries for the London stock in order to investigate in greater depth the distribution of $PM_{2.5}$ concentrations and the impacts of a variety of interventions and archetypes. The scenario studied in this chapter is, as acknowledged, an extreme one aiming to show the upper boundary of what could possibly be achieved. In addition, a comparison with other locations are needed to consider if results in relation other towns/building stocks are different and whether those shown here are location specific. This is the focus of the next chapter. Comparisons with published empirical and modelling studies confirmed the effective use of the modelling techniques for pollution investigation. Further data is needed in order to carry out a more thorough sensitivity analysis, especially for those variables shown in Table 5.10 where some values have had to be assumed due to lack of data, and also for those which show the greatest impact on indoor $PM_{2.5}$ concentrations.

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Chapter 6

Variation in Indoor $PM_{2.5}$ Exposure Between Locations in England: London and Milton Keynes, a More Realistic Case

Introduction

The previous chapter explored the application of multi-zonal modelling software using a specific location (London) and a specific energy efficiency/ventilation scenario (MVHR with airtightness reducing permeability to $3m^{-3}m^{-2}\ hr^{-1}$ at 50Pa for all properties) to quantify changes in $PM_{2.5}$ concentrations. Having shown this to be an effective method for modelling change when compared to both monitoring and other modelling studies, this chapter expands the scope of the investigation, by applying a wider (and perhaps more realistic) range of energy efficiency interventions and further building archetypes in order to explore possible differences in indoor exposure to $PM_{2.5}$ between two contrasting English locations - London and Milton Keynes - in greater depth. It considers what the contributions are to the targeted reductions in CO_2 emissions required by the Climate Change Act (2008) as a result of the application of such measures and the impact on indoor $PM_{2.5}$ concentrations when purpose provided ventilation (PPV) strategies in accordance with ADF, 2010 are included as part of the package and when they are not. Additionally, the updated modelling is now combined within the SCRIBE tool (discussed below) and enables models to calculate end-used energy demand when run against a variety of possible grid decarbonisation scenarios. This represents a more realistic scenario that combines a number of policy options and the joint goals of reducing emissions whilst maintaining good indoor air quality (IAQ). Although not the focus of this thesis, since the work was carried out as part of the PURGE project WP4 (PURGE, 2012); moisture (a precursor for mould), environmental tobacco smoke (ETS) and radon were also investigated. As $PM_{2.5}$ is one of many airborne indoor pollutants, the results provide insight in the selection of optimal strategies for health by comparing results for various pollutants alongside each other. The investigations in this chapter resulted in a peer reviewed research paper (Shrubsole et al., 2015) as well as a related conference paper. Additionally, much of background to chapter 6 is contained in the report 'Summary for Policy Makers and Dissemination Guidelines for the EU' (PURGE, 2013, 2014) for which the author was a main contributor and editor.

The UN Report: Cities and Climate Change: Global Report on Human Settlements, 2011 commented that future urban climate mitigation efforts were not likely to be regionally differentiated, but rather characterized by differences between an elite group of cities with access to substantial resources, primarily in developed countries, and the vast majority of cities for whom addressing climate change would remain a low priority (UN, 2011). This points to the need for comparison between diverse case study cities. As a consequence, London was chosen as an 'already large city' and Milton Keynes as a 'small but growing' city for the UK element of PURGE project (PURGE, 2012). In addition, the proportion of archetypes and existing levels of energy efficiency varies between the two cities (Summerfield et al., 2007), with London having a greater range of older properties, as opposed to Milton Keynes where the majority of properties were built post 1965 (MKiO, 2014).

London is a major city with an estimated 2010 population of 7.83 million, responsible for 8.4% (44.71Mt) (GLA, 2010) of UK CO_2 emissions, and characterized by both new and old buildings and higher density forms; Milton Keynes is a town 72 km north-west of London, created under the UK's 2nd New Towns Act 1965, with an estimated 2010 population of 241,500, and responsible for 0.3% (1.76 Mt) of UK CO_2 emissions (MKiO, 2014), with predominantly more recent and low-density building archetypes. Population data from 2010 is used as this represents the start date for the scenarios that follow. For each location the existing (2010) housing stock is modelled with three future scenarios with different levels of energy efficiency interventions combined with either a business-as-usual, or accelerated decarbonization of the electricity supply grid approach. A greater range of building archetypes, energy efficiency and ventilation measures are used compared to the previous chapter in order to be more representative of locations and to allow a more comprehensive investigation. CONTAM is then used to create the simulations within SCRIBE -a building physics-based health impact model of the UK housing stock linked to the English Housing Survey - in order to examine changes, 2010-2050, in end-use energy demand, CO_2 emissions, winter indoor temperatures, airborne pollutant concentrations and associated health impacts (Hamilton et al., 2015).

Results from this research were published in a peer-reviewed journal (Shrubsole et al., 2015) and contributed to other papers as detailed under 'thesis associated publications' beginning at page 14.

6.1 Methodological Approaches

6.1.1 Background to Scenario Modelling

National targets for CO_2 emissions reduction in the UK, which are one of the main drivers of changes in energy performance in dwellings (Rosenow, 2012), are set out in the Climate Change Act of 2008 (HM Government, 2008). Individual sectors such as housing, have contributory targets (CCC, 2012). Total emission reductions from the housing stock will occur both through energy efficiency interventions and by decarbonizing the dwelling energy supply. The carbon intensity (CI) of the supply grid will influence future CO_2 emissions depending on the mix of sources, e.g. coal or gas fired power stations, renewables and nuclear. Carbon intensities are expressed in grams of CO_2 per kWh and are estimates of equivalent CO_2 emissions normalized per unit of delivered electricity (i.e. including transmission and distribution losses). A number of scenarios have been described for decarbonizing the grid, in line with emissions reductions targets (CCC 2010). These changes to power generation, in conjunction with policies aimed at transport and industry are also expected to reduce airborne pollution, improving future air quality (Williams, 2007). The coupling of fuel source and power grid decarbonization scenarios with energy efficiency retrofits to the housing stock and the impact on IAQ and health is a relatively new area of research. To date few studies have investigated how such interactions vary between settings with differing housing stocks (Hamilton et al., 2015), although this is increasingly recognized as a factor in achieving UK wide CO_2 reduction targets by Government (DECC, 2013).

6.1.2. Modelled Scenarios: Decarbonization of the Housing Stock and Electrical Grid

This chapter models the current stock (2010) in both locations and the impact of three future housing/electricity grid decarbonization scenarios applied to the housing stock in London and Milton Keynes (Table 6.1).

Table 6.1 Combined future scenarios for grid decarbonisation and housing energy interventions

Grid Decarbonisation Scenario	Range of Energy Efficiency and Ventilation Interventions[1]	Source	CI[2] 2010	CI 2020	CI 2030	CI 2050
1. Energy Efficiency: A range of energy efficiency and ventilation housing interventions with no decarbonisation of the electricity grid.	Draught Stripping New Double Glazing with Trickle Vents Extract Fans Cavity Wall Filling Solid Wall Insulation Insulate Lofts to 250mm[3]	UKERC (2013)	464	480	420	360
2. Energy Efficiency Plus: A greater range of energy efficiency and ventilation housing interventions occur with no decarbonisation of the grid.	Draught Stripping New Double Glazing with Trickle Vents Extract Fans Cavity Wall Filling Solid Wall Insulation Insulate Lofts to 250mm Install Condensing Boilers Central Heating	UKERC (2013)	464	480	420	360
3. Low Carbon Supply: An ambitious scenario; a range of energy efficiency and ventilation housing interventions: (1. Energy Efficiency) occur at the early stages (2010-2020) to reach UK interim targets. Major decarbonisation of the electricity grid.	Draught Stripping New Double Glazing with Trickle Vents Extract Fans Cavity Wall Filling Solid Wall Insulation Insulate Lofts to 250mm	UKERC (2013)	464	290	70	25

[1]Energy efficiency and ventilation interventions are applied to those houses not having them according to the English Housing Survey (EHS 2010).
[2]CI (carbon intensities) expressed in grams of CO_2 per kWh. These are estimates of equivalent CO_2 emissions normalized per unit of delivered electricity (i.e. including transmission and distribution losses).
[3]Loft insulation topped up to or installed to 250mm (BRE, 2009)
Arrow denotes direction of increasingly aggressive supply decarbonisation scenarios

Starting from the 2010 stock in both locations, interventions were applied that brought the 2050 stocks to parity in order to fairly quantify and compare changes in indoor $PM_{2.5}$ concentrations and other airborne pollutants and the possible health impacts, energy use and CO_2 savings. The future contrasting scenarios are:

(i) 'Energy Efficient (EE)': This assumes a business-as-usual trajectory with regard to the carbon intensity of the electricity grid to 2050 with a range of housing interventions applied to all those properties not currently having them. For housing interventions, data on the existing measures in the current stock (2010) were derived from a variety of empirical sources (EHS, 2012; CSE, 2012; HECA, 2013; HEED, 2014; MKiO, 2014).

(ii) 'Energy Efficiency Plus (EE+)': This assumes business-as-usual carbon intensity of the grid, but with additional housing energy efficiency interventions focused on heating and seeks to investigate the impact of a greater focus on technical adaptation of dwellings. Again, these are applied to all properties not currently having them and therefore represent the upper bound case.

(iii) 'Low Carbon Supply (LCS)': This assumes an aggressive supply decarbonization scenario with housing interventions as in (i) and that space heating in houses will be 100% electrified by 2050.

All scenarios and their individual components start from a base line of 2010 and were specified to coincide with the CO_2 reduction target date of 2050.

The grid decarbonization scenario used in (i) and (ii) are equivalent to the 'resilient' scenario, whilst (iii) is equivalent to the 'Low-carbon' scenario both seen in the UKERC Research Report (UKERC, 2013).

The baseline (2010) figure for carbon intensity (CI) in the scenarios comes from data for centralized electricity generation from the Digest of UK Energy Statistics, DUKES (2011). The emission reductions of the energy supply grid and power sector fuel mix and grid emission figures for carbon intensity (CI) were derived from the UK Energy Research Centre (UKERC) scenarios within the UK Committee on Climate Change 4th Carbon Budget report. (CCC, 2010; UKERC, 2013). These were chosen as they allow for structural uncertainties in future energy supply and represent the upper and lower boundaries of possible grid decarbonization. The year by year CI figures represent national targets with local trends assumed to evolve similarly over time. UKERC scenarios reduce grid emissions through specific investment choices, such that remaining sector reductions (including housing) are to be achieved through technology, efficiency and conservation (UKERC, 2013).

It is assumed that all installed measures are replaced once their life expectancy is over (e.g. boilers are replaced after 15 years). Due to uncertainties, no allowance is made for possible future improvements in efficiency or new technology. Population figures for 2010 are used to coincide with scenario start dates and inform health calculations.

Although the UK building regulations require that the air quality is made no worse following retrofitting, there is no specific guidance around ventilation for energy efficiency retrofits. All future scenarios were run specifying the inclusion of purpose-provided ventilation (PPV) (extract fans and trickle vents) to maintain adequate air exchange following airtightening in accordance with current Building Regulation requirements for new builds (HM Government, 2010). Given the potential importance that ventilation has on air quality in the home (Bone et al., 2010), additional simulations without PPV were run, so as to examine the importance of ventilation characteristics for impacts on CO_2 emissions and health.

6.1.3 Modelling the Scenario Impacts: SCRIBE

Modelling of scenario impacts on CI, health, and mortality was carried out using a UK housing stock computer model known as SCRIBE (Strategies for Carbon Reduction In the Built Environment), developed by University College London (UCL) and the London School of Hygiene and Tropical Medicine (LSHTM). SCRIBE is a variant of the HiDEEM tool (Hamilton et al, 2015), that includes variable external $PM_{2.5}$ concentrations, and carbon intensities (CI) for the grid. SCRIBE is a unique tool in that it integrates a variety of different individually validated components/modules by linking these through a software environment developed in the 'R' programming language (R, 2017). Although, Hamilton et al, (2015) explores the natures of the tool and its use, this section of the thesis, applies the expanded to tool investigate a specific set of scenarios and locations. SCRIBE incorporates (i) a building physics module that enables estimation of energy use, indoor environmental conditions (winter temperatures and annual pollutant concentrations) and CO_2 emissions under a range of housing interventions and projected changes in grid carbon intensities, and (ii) a model of health impacts associated with indoor environmental conditions. Modelled CO_2 emission reductions are compared to emission levels required to meet targets set in the Climate Change Act 2008, from a base line of 2010, rather than 1990 (HM Government, 2008). The baseline of 2010 is used due to limitations in data availability for some SCRIBE inputs, particularly building/intervention data for Milton Keynes prior to this date. Consequently, CO_2 emission reductions targets are adjusted as follows: for 2020-43%, for 2030-57% and for 2050-75%, all relative to 2010 instead of 1990. This adjustment has no impact on the 2050 results. Details of the SCRIBE model and the inputs used in the various components are outlined in Figure 6.1

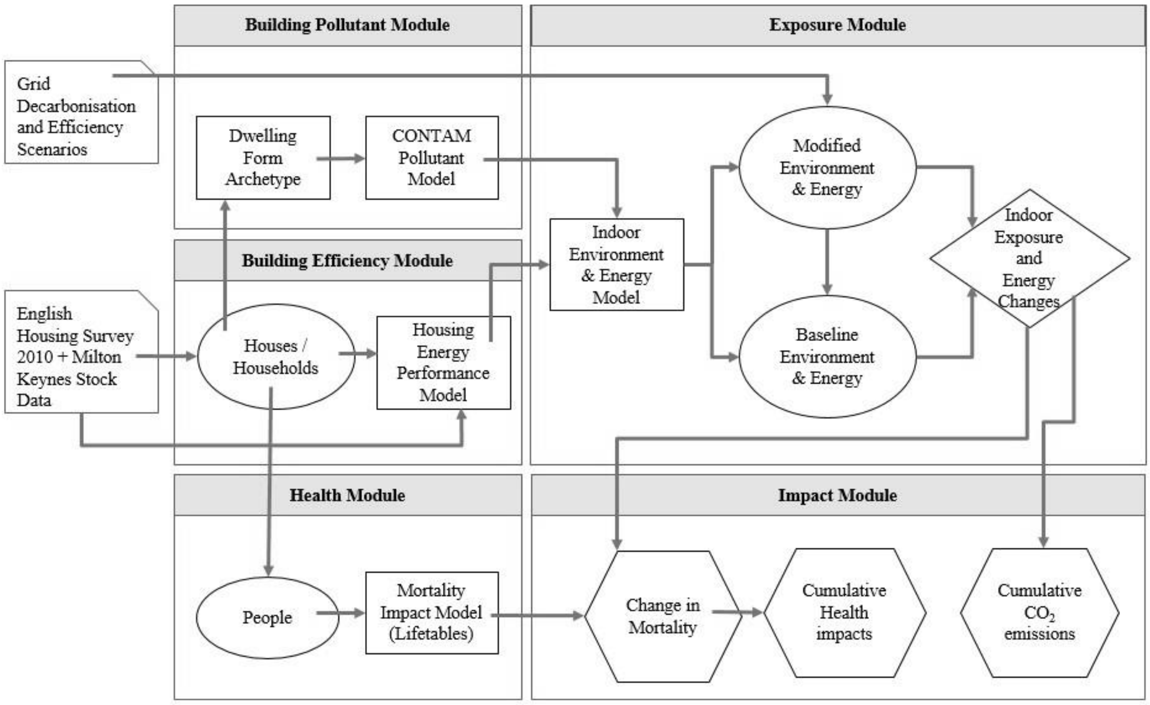

Figure 6.1 Connections between grid decarbonisation and energy efficiency and ventilation measures in housing and the impacts on health and CO_2 emissions within the SCRIBE tool (adapted from Hamilton et al., 2015). The archetypes and their variants were constructed by the author in CONTAM 3.1 and the pollutants added.

6.1.4 Building Pollutant Module

As per the principles outlined in Chapter 3, the building pollutant module within the SCRIBE tool contains inputs produced using the CONTAM models updated to version 3.1.

6.1.4.1 Built Forms

Valuation Office Agency data VOA (2014) was used in conjunction with data from the Census data (Census, 2011) and the Milton Keynes observatory (MKiO, 2014), to produce a profile of dwelling types for Milton Keynes. The resultant built forms are matched to each EHS entry using criteria of dwelling type and size. For London data was drawn directly from the EHS data base. Archetypes representative of the London and Milton Keynes housing stocks were constructed in CONTAM by the author to assess changes to the indoor environment (air quality, winter temperature and energy use) associated with the various interventions applied to dwellings in the 2010 English Housing Survey (EHS, 2012). Ten geometries were used for analysis of the domestic stock in each location. Nine were derived from the LUCID project (Oikonomou et al., 2012). The remaining example, House 7, was taken from the Lancet study (Wilkinson et al., 2009), supplemented with typical floor plans and facades available from the literature. Building geometries are shown in Chapter 4 table 4.1. These archetypes are considered to be representative of the housing stocks of London and Milton Keynes based on the frequency of their occurrence according to Census, VOA, MKiO and EHS data. The full details of all geometries are shown in Appendix C. These archetypes represent a natural progression from those used in chapter 4, by offering a wider range of geometries, energy efficiency and ventilation measures, and are being linked to the EHS, providing a more robust basis for investigation

For baseline (2010) indoor pollutant concentrations, each geometry in CONTAM is remodelled with four distinct ventilation system options: (i) no trickle vents or extract fans, (ii) trickle vents only, (iii) extract fans only and (iv) trickle vents and extract fans. This gives a total of 40 dwelling form-ventilation archetypes with which to represent the EHS dwelling base types and variants (those with additional ventilation options). All ventilation components are assumed to be functioning correctly with no allowance made for mechanical failure or deterioration with time. It is acknowledged that this could lead to slightly lower indoor pollutant concentrations resulting from modelling. Each of the 40 archetypes is modelled with eight permeabilities ranging from 3 to 30 $m^3/h/m^2$@50Pa (Steven 2000), giving a total of 320 possible archetypes.

6.1.4.2 Building Fabric and Ventilation

For London, existing energy and ventilation interventions are modelled directly from the English Housing Survey (EHS, 2012), which comprises a representative sample of properties (16,150 surveyed dwellings) with weights for each dwelling variant which can be used to represent all households in England. Government Office Region (GOR) information enables London dwelling variants to be directly selected from the EHS and used in the modelling. The survey does not have a sufficient or identifiable sample for Milton Keynes, which is one of a number of urban conurbations in the South East of England. Dwellings and their distribution and frequency in each location were therefore simulated by using alternative empirical data sets (researched by the author) for the variables required for SCRIBE. For example, the range of energy efficiency installations that have been implemented in the domestic building stock (CSE, 2012; HECA, 2013) and HEED (2014), which also contains combined Energy Performance Certificate and Energy Saving Trust Home Energy Check data.

Dwelling age, type and distribution were sourced by the author from the Milton Keynes Observatory data base (MKiO, 2014). For the few remaining variables that were not available: (i) the Standard Assessment Procedure[2] (SAP) rating, (ii) envelope permeability and (iii) ventilation type; the known variables were used to calculate estimates for SAP rating and envelope permeability. Using an algorithm supplied by Dr Payel Das (UCL) and implemented in the R programming language (R, 2017) Latin hypercube sampling (LHS) was used to calculate the probability of occurrence of the variables in each of the ~16,000 EHS variants and were then used to randomly sample the Milton Keynes housing stock and scaled to the correct number of dwellings. The stock modelling input variables and their ranges are shown in Table 6.2.

[2] SAP: The Government's Standard Assessment Procedure for Energy Rating of Dwellings (BRE, 2012)

Table 6.2 Stock modelling input variables and their ranges

Stock Variables	Variable Range	London Stock: Source	Milton Keynes Stock: Source
Dwelling Types	End terrace, mid terrace, semi-detached, detached, bungalow, converted flat, purpose built flat-low rise, purpose built flats-high rise	English Housing Survey (EHS, 2012)	2011 Census; MKiO, 2014
Dwelling Age	Pre 1919, 1919-44, 1945-64,1965-80, 1981-90, post 1990		2011 Census; MKiO, 2014
Wall Types	Cavity with insulation[1], cavity uninsulated, solid uninsulated[2]		CSE, 2012
Glazing Types	mixed[3] single[4] double		CSE, 2012; MKiO, 2014
Eligible for Loft Insulation	Yes, No[1]		CSE, 2012; HEED, 2014; MKiO, 2014
Eligible for Boiler Upgrade	Yes, No		CSE, 2012; MKiO, 2014
Eligible for Central Heating Upgrade	Yes, No		CSE, 2012; HECA, 2013; MKiO, 2014
Ventilation Type	No trickle vents or extract fans, trickle vents only, extract fans only trickle vents and extract fans		EHS, 2012[5] and allocated as under section 6.1.4.2
SAP Level	<30, 30-50, 51-70, >70		EHS, 2012[5] and allocated as under section 6.1.4.2
Permeability@ 50Pa	3, 5, 7, 10,15, 20, 25, 30	Distribution: Stephens, 1998, 2000	EHS, 2012[5] and allocated as under section 6.1.4.2

[1]likely slightly underestimates totals for Milton Keynes as does not include properties receiving measures prior to 2005 as no reliable data exists (CSE, 2012).
[2]assumes all solid wall properties have the potential for insulation
[3]lack of data for mixed types
[4]assumed if not double glazed
[5]as no data available, weighting for South East region used

6.1.4.3 Contaminants: Emission Rates and Schedules

Each one of the CONTAM 320 models is mapped to a particular EHS variant using the predicted permeability value. These are simulated in CONTAM, to obtain concentrations of a number of pollutants: indoor and outdoor sourced particulate matter ≤2.5µm ($PM_{2.5}$), radon, environmental tobacco smoke (ETS), and moisture (as a precursor of mould). Under each scenario the adapted EHS variants are mapped to the CONTAM models to reflect the change in permeability following the interventions, thus future changes in indoor pollutant concentrations and energy use are estimated. This mapping includes anticipated reductions in external $PM_{2.5}$ concentrations, specified by year and location. For London, the 2010 annual mean outdoor urban background concentration $PM_{2.5}$ is taken as 13.0µg.m^{-3}

(Shrubsole et al., 2012). For Milton Keynes, the figure of 10.9 µg.m^{-3} is based on data from the DEFRA mapping project (DEFRA, 2013). For future $PM_{2.5}$ concentrations, DEFRA data is used up to 2030 in both locations. For 2050, in the absence of further data, a linear trend based on 2010-2030 data is assumed and accounts for changes in outdoor $PM_{2.5}$ concentrations with time in the various grid decarbonisation scenarios. The SCRIBE model enables differentiation between $PM_{2.5}$ from indoor and outdoor sources due to the ongoing uncertainty regarding the toxicity of particles generated indoors (Hamilton et al., 2015).

Radon exposures are informed by the national distribution reported in Gray (2009) and adapted to allow for regional differences in emission rates (HPA, 2011). Smoking levels for each location are informed by NHS, 2011. No account is taken of any possible future changes in smoking prevalence, or indoor/outdoor smoking behaviour that may influence exposure for non-smokers in smoking households. Indoor pollution emission profiles are derived from empirical studies (Table 6.3).

For outdoor conditions influencing indoor values, transient yearly weather files are constructed using Chartered Institution of Building Services Engineers (CIBSE) Test Reference Year (TRY) and Design Summer Year (DSY) data.

Table 6.3 Data sources for indoor pollutant inputs

Pollutant	Values	Data Source
$PM_{2.5}$ emission* $PM_{2.5}$ deposition*	Cooking 1.6mg/min 0.39l/h	Ozkaynak *et al.* (1996), Chen and Zhao (2011)
Radon	0.005 Bq, 0.05 Bq and0.1Bq (one decay s^{-1}) x's room floor area m^2	Fang and Persily (1995), Gray (2009) , HPA (2011)
Environmental Tobacco Smoke^	0.99 mg/min at 5 minutes per cigarette	NHS (2011), He et al. (2004), Afshari et al. (2005)
Moisture (precursor of mould)	Various values depending on source (n=7)	FMNectar (2007), Gilbertson et al. (2012)

*the emission rates seen here are the same used throughout the thesis as are all the schedules that accompany them.
^In this scenario ETS is treated as a separate pollutant in line with the requirements from BEIS (formally Department of Energy and Climate change), who commissioned the HiDEEM tool.

Within the CONTAM modelling, each pollutant has a defined source and emission period: indoor $PM_{2.5}$ is a function of occupancy and cooking; moisture is a function of occupancy and bathroom use; and ETS is a function of occupancy, with weekend and weekday occupancy profiles differing. These include:

- **$PM_{2.5}$: Particulate Matter with Aerodynamic Diameter 2·5µm or Less**

Internally generated $PM_{2.5}$ comes from cooking in the kitchen. It was modelled as per chapter 4, section 4.1.2, with a generation rate of 1.6mg/min and a fixed continuous deposition rate in all rooms of 0.39l/h (Ozkaynak *et al*. 1996). The cooking schedules can be seen in (chapter 4, table 4.5), representing assumed normal behaviour. A no-internal source scenario was run to distinguish between externally and internally generated $PM_{2.5}$.

- **Environmental Tobacco Smoke (ETS)**

ETS exposure is simulated using a tracer gas model which simulates 1 smoker, smoking the average number of cigarettes per person per day (14 cigarettes per day for men and 13 cigarettes per day for women – based on UK averages (NHS, 2011). Given that people also smoke outside of their property; and in the absence of precise data, we assumed that 2 cigarettes were smoked in the kitchen on weekdays and weekends and 4 cigarettes on weekdays and 7 at weekends in the living room using data from Wilkinson et al. (2009). An emission rate of 0.99 mg.min-1 at 5 minutes per cigarette was used based on both residential measurements and chamber experiments (He et al. 2004; Afshari et al. 2005). There is no deposition rate specified for this pollutant. Units are dimensionless.

- **Radon**

All models are run at a low fixed continuous exposure source of 0.005 Bq (one decay per second) (Fang and Persily 1995), with a source point in each ground floor room. The multiplier function in CONTAM is used calculate the room emission based on the floor area. Calculations were made for two other exposures; medium 0.05 Bq and high 0.1Bq in order to represent all the concentration bands seen in the UK (Gray et al, 2009).

- **Moisture (Precursor of Mould)**

Using the moisture rates seen in Table 6.4 and hourly schedules based on the occurrence of activities, number of persons and their location (Tables 6.5 and 6.6, both from FMNectar (2007), an overall moisture content was calculated for each relevant room, which was reflected in the room scheduling in CONTAM for moisture.

Table 6.4 Moisture generation rates

Household activity: moisture generation rate	
Activity (per person)	Rate
Sleeping	40 g/h per person
Active	55 g/h per person
Cooking: Electric	2,000 g/day
Cooking: Gas	3,000 g/day
Dishwashing	400 g/day
Bathing/washing	200 g/person per day
Washing clothes	500 g/day
Drying clothes indoor (e.g. unvented tumble dryer)	1,500 g/person per day

Table 6.5 Moisture generation per activity per hour

Moisture generation per activity in grams per hour																								
Time	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24
Number of people	3	3	3	3	3	3	3	3	3	1	1	1	1	1	1	1	2	2	2	3	3	3	3	3
Moisture gen. rate /person	40	40	40	40	40	40	40	55	55	55	55	55	55	55	55	55	55	55	55	55	55	55	40	40
Tot. moisture - people	120	120	120	120	120	120	120	165	165	55	55	55	55	55	55	55	110	110	110	165	165	165	120	120
Cooking									600					600					1000	1000				
Bathing								400													200			
Dishwashing									200												200			
Washing clothes											250	250												
Drying Clothes indoors													250	200	150	100	100	100	100	100	100	100	100	100
Plants	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10
Total. moisture (g)	130	130	130	130	130	130	130	575	975	65	315	315	315	865	215	165	220	220	1220	1275	675	275	230	230

Table 6.6 Moisture generation per relevant room

Moisture generation per relevant room in grams per hour (based on average 3 Person household)																								
Time	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24
Bed 1	80	80	80	80	80	80	80																80	80
Bed 2	40	40	40	40	40	40	40																40	40
Bathroom								400	0	0	0	0	83	67	50	33	33	33	33	33	233	33	33	33
Living room	10	10	10	10	10	10	10	93	93	38	38	38	121	104	88	71	98	98	98	126	126	126	43	43
Kitchen								83	883	28	278	278	111	694	78	61	88	88	1088	1116	316	116	33	33
Tot. moisture (g)	130	130	130	130	130	130	130	575	975	65	315	315	315	865	215	165	220	220	1220	1275	675	275	230	230

6.1.4.4. Post-Processing of Contaminant Outputs

Modelling and post-processing of pollutant concentrations vary in methodology as follows, but all occur automatically with SCRIBE tool. Annual average concentrations were calculated for each archetype and variant by combining summer and winter versions of each file by weighting (0.33xSummer) which represents 4 months + (0.67xWinter) representing 8 months following the methodology in Wilkinson *et al.* (2009). For all contaminants (except moisture/mould, - see below), output exposures are initially manipulated to reflect the household average based on the weighted average annual concentration in the living room, bedroom 1 (defined as the largest/master bedroom in the property) and the kitchen, using time weighting factors of 0.45, 0.45, and 0.1, respectively based on their importance in terms of occupation and the assumed proportions of time spent in these rooms by the average occupant (Wilkinson *et al.* 2009). Results, once post processed were used to populate the SCRIBE tool for each archetype, ventilation scenario and permeability.

- ***ETS****:* The annual average ETS concentration is scaled by the average value for the stock based on the EHS data.
- ***PM$_{2.5}$****:* The results are not post-processed any further.
- ***Radon****:* Results are further post processed to produce a single value for each geometry by applying the UK distribution where 90% of houses are low exposure (le), 9% are medium (me) and 1% are high (he) (Gray et al., 2009). For Flats, a further computation is required for the number of Flats on the first floor and above. Data from ONS sources show 40% at ground floor (full exposure), 14% at first floor (50%) exposure and the remainder above at 0 exposure. 1. UK Average exposure for House models = [(0.9 x le) + (0.09 x me) + (0.01 x he)] 2. UK Average exposure for Flat models = [(0.9 x le) + (0.09 x me) + (0.01x he)]. The result is adjusted for variation in flat heights = [(answer equation 2 x 0.4) + ({answer equation 2 x 0.5} x 0.14)]
- ***Moisture****:* Mould Severity Index (MSI): The hourly values of the humidity ratio in the living room during the winter months were used to calculate the MSI. The internal humidity ratio is converted into an internal vapour pressure and the hourly excess of the internal vapour pressure over the external vapour pressure (as given in the winter weather file) is then calculated. The internal vapour pressure excess (VPX) is then regressed with the external temperature to estimate VPX at an external temperature of 5˚C, VPX5C. The external saturated vapour pressure is calculated at a temperature of 5˚C (eSVP5C) and internal saturated vapour pressure is calculated at a temperature estimated from the permeability using Warm Front data (iSVPwf). The standardized internal relative humidity at an external temperature of 5˚C and external relative humidity of 80% is then calculated (SRH). The % of dwellings for each geometry and permeability with MSI > 1 is finally estimated using the empirical relation with SRH found in the Warm Front data for living rooms (Oreszczyn et al. 2006)

6.1.5. Building Efficiency Module

The building efficiency module estimates envelope permeability and heat loss resulting from fabric performance, heating system and ventilation characteristics. A conversion process uses EHS variables to infer features, such as dwelling geometry and construction characteristics to predict ventilation and thermal performance (DECC, 2012). The SAP criteria is then used to predict total ventilation rate, dwelling permeability, and fabric heat loss rate (Hughes et al., 2013). These are combined with the heating system performance to predict a heat transfer characteristic E-value[3] for each dwelling, using a relationship that takes into account the expected behaviour of the occupant (Oreszczyn et al, 2006). Each intervention is associated with changes in the thermal and ventilation characteristics of the EHS variants. The SAP method is used to calculate the new heat transfer characteristic, and the new E-value is predicted such that changes in energy use can be estimated under the variety of future scenarios. For ventilation changes, the application of an intervention results in a change in the EHS variant to which the updated geometry is matched. The permeability of this 'new' EHS variant is assumed to represent the ventilation change occurring post intervention. These include PPV, designed to comply with Approved Document F1 (HM Government, 2010). One of the key assumptions in the health impact modelling (section 5.2.4) is that additional PPV will be installed in dwellings alongside the energy efficiency measures.

6.1.6 Health Impacts Module

The health impacts associated with the changes in indoor exposure, were calculated within the health module, which was created by Dr James Milner at LSHTM as part of the HIDEEM/PURGE project. It is described here to give background and insight into the process as part of the investigation into the impacts of variations in $PM_{2.5}$ concentrations as a consequences of the application of energy efficiency and ventilation interventions. The health impacts associated with changes to annual indoor air quality and heating season temperatures, were modelled within the SCRIBE tool using life table methods based on the IOMLIFET model (Miller and Hurley, 2003) using all-cause and cause-specific mortality data for England and Wales available from the Office for National Statistics (ONS), with separate life tables for males and females. The key model output was changes in years of life lived as a result of changes in mortality risk. Exposure-response relationships for changes in indoor exposures (i.e. standardized internal temperature (SIT), ETS, $PM_{2.5}$ derived from indoor and outdoor sources and radon) were derived from published sources shown in Table 6.7 and were assumed to be log-linear with no thresholds. Where more than one exposure was related to the same health outcome (e.g. lung cancer mortality from both $PM_{2.5}$ and radon, it is assumed that the relative risks are multiplicative in line with the work of Scarborough et al. (2010).

[3] The E-value represents the dwelling heat transfer characteristic, obtained by combining the estimated fabric and ventilation performance with the heating system (after Oreszczyn et al, 2006).

Table 6.7 Modelled mortality outcomes and exposure-response relationships

Exposure[1]	Health outcome	Exposure-response relationship[2]	Reference
Standardized internal temperature (SIT)	Winter excess cardiovascular mortality	0.98 per 1°C[2]	Derived from Wilkinson et al. (2001)
Environmental tobacco smoke	Cerebrovascular accident (stroke) mortality	1.25 (if in same dwelling as smoker)	Lee and Forey (2006)
	Myocardial infarction (heart attack) mortality	1.30 (if in same dwelling as smoker)	Law et al. (1997)
$PM_{2.5}$	Cardiopulmonary mortality	1.082 per 10 µg/m^3	Pope et al. (2002; 2004)
	Lung cancer mortality	1.059 per 10 µg/m^3	As above
Radon	Lung cancer mortality	1.16 per 100 Bq/m^3	Darby et al. (2005)

[1]Mould/moisture is not included as while there is a relationship to morbidity, no clear relationship with mortality (the object of this study currently exists).
[2]exposure-response relationships vary in terms of the shape of their relationship. For example, in the case of SIT there is a threshold value below which the relationship applies, whereas for radon the relationship is linear.
[3]The mortality risk decreases by 2% for every 1°C increase in winter temperature (SIT).

Health impacts were modelled year by year to 2050. For all modelled outcomes other than those associated with changes in SIT, we specified outcome-specific inception and cessation lag functions to reflect the time delay between changes in exposure and subsequent change in disease status using data from Hamilton et al. (2015).

6.2 Results

6.2.1 Energy Use and CO_2 Emissions

Changes in Energy consumption (kWh) and CO_2 emissions of the housing stocks relative to the 2010 baseline taking account of the changing carbon intensity (CI) of the electrical grid are shown for each scenario in Table 6.8. Values are expressed as the percentage increase relative to the base year of 2010, with negative figures therefore indicating reduction in CO_2 emissions.

Table 6.8 Changes in mean *per capita* energy consumption and CO_2 emissions of the London and Milton Keynes housing stock under each of the scenarios with and without purpose-provided ventilation. Negative values signify reduction in energy or CO_2 emissions compared with 2010.

Location	Scenario *	% Change in mean energy use 2010-2050		% Change in mean CO_2 emissions 2010-2050	
		With purpose-provided ventilation	Without purpose-provided ventilation interventions	With purpose-provided ventilation	Without purpose-provided ventilation interventions
London	EE	- 37.7	- 35.5	- 50.0	- 51.7
	EE+	- 44.9	- 42.9	- 55.7	- 57.3
	LSC	- 37.7	- 35.5	- 96.6	- 96.7
Milton Keynes	EE	- 16.7	- 14.9	- 33.9	- 35.3
	EE+	- 26.6	- 24.9	- 41.7	- 43.0
	LCS	- 16.9	- 14.9	- 95.6	- 96.1

*See table 6.1 for explanation of scenarios.

Greater reductions in energy use are seen in London relative to Milton Keynes under all intervention scenarios, with the highest gains seen in the EE+ scenario where no additional purpose provided ventilation (PPV) was assumed. This is likely due to the older less efficient stock profile in London. For both the EE and EE + scenarios, appreciably greater CO_2 reductions were seen in London than in Milton Keynes. Aggressive decarbonization of the electric grid, combined with housing measures in the LCS scenario exceeded the targets needed for compliance with the Climate Change Act, 2008 in both locations. The addition of ventilation interventions increased energy use by an average of 8.6% across the scenarios.

6.2.2 Temperature and Pollutant Concentrations Changes

Table 6.9 shows the changes that occur in mean indoor temperature during the heating season and annual airborne pollutant concentrations following the installation of both energy efficiency and PPV interventions under the three scenarios for 2050.

Table 6.9 Mean indoor pollutant concentrations and temperatures, 2010 and 2050, for each of the scenarios with/ without purpose-provided ventilation.

EE and LCS Scenarios	London			Milton Keynes		
	Current	With PPV	Without PPV	Current	With PPV	Without PPV
Exposures to	**2010**	**2050**	**2050**	**2010**	**2050**	**2050**
Standardized indoor temperature (SIT, °C)	17.7	18.0	18.2	17.9	18.2	18.3
ETS	1.0	0.9	2.3	1.0	0.9	1.7
Indoor $PM_{2.5}$ derived from outdoor sources (μg/m^3)	5.5	4.0	2.8	5.6	3.8	3.1
Indoor $PM_{2.5}$ derived from indoor sources (μg/m^3)	12.5	5.1	16.1	9.5	4.1	11.8
Radon (Bq/m^3)	14.0	11.8	32.8	28.2	26.4	55.1
Mould (% with mould index >1)	17.2	10.1	25.2	9.2	7.1	12.9
EE+ Scenario*						
Standardized indoor temperature (SIT, °C)	17.7	18.1	18.3	17.9	18.3	18.4
Mould (% with mould index >1)	17.2	9.9	25.0	9.2	6.9	12.8

*Impacts for ETS, $PM_{2.5}$ and Radon remain constant

There are some items of note in Table 6.10 that require explanation. Although the Milton Keynes stock is generally more energy efficient, as can be inferred from the slight difference in current standardized indoor temperature, this does not mean it is necessarily more airtight. The prevalence of semi-detached and detached properties is likely to lead to greater airchange rates due to more external surfaces than for example flats in London. This may explain why current-2010-concentrations of indoor $PM_{2.5}$ derived from outdoor sources are similar, despite the lower outdoor concentrations for Milton Keynes. In contrast, indoor $PM_{2.5}$ derived from indoor sources, is higher in London. It is likely that whilst airchange rates also play a part the smaller mean geometries seen in London are the major factor here.

It is acknowledged that the mean indoor concentrations from both external and internal $PM_{2.5}$ for London (18.1μg/m^3) varies in comparison to the figure from chapter 4 table 4.7 (28.4μg/m^3). However, the previous chapter considered one scenario: airtightening and MVHR for all properties in London using only two baseline archetypes to represent the domestic stock. This chapter considers multiple geometries, the changes in permeability resulting from the addition of a number of measures. All other schedules (occupancy, ventilation behaviour, periods and rates of pollutant emission etc.) were held constant between the studies. The figure from this study is arrived at by using a greater range of ventilation and energy efficiency strategies on a larger range of geometries linked to the English Housing Survey's 16,000 geometries and therefore seen as more reliable for the general mean and distribution of concentrations and are still within the range of values for indoor $PM_{2.5}$ seen in available empirical and modelling studies. The ingress of $PM_{2.5}$ derived from the outdoor air was reduced by the greater airtightness without PPV, but concentrations of all other pollutants showed increases from the 2010 base line for both locations. London dwellings suffer more, due to greater relative reduction in envelope permeability and therefore air-tightness. The exception is radon because of the greater emission levels seen in Milton Keynes (which are determined by local geology) and the more airtight nature of the initial (2010) building stock (HPA, 2011). A mould risk of >1 indicates the likely presence of mould in a property, with figures representing the % of properties where the mould risk is >1. A decrease shows a reduction in health risk. However, these changes in mould risk have not been used in the calculation of health impact for this work (though there is some evidence of likely impact, particularly in children) as this study is only reporting mortality impacts and not morbidity. The changes in mean indoor heating season temperatures were only marginally greater without PPV than in the scenarios which included it. For scenarios with PPV higher mean indoor temperatures during the heating season are seen in the housing stock following retrofitting, with appreciable reductions in most of the pollutants studied except for radon gas, which shows small reductions in both locations. As a continuous source radon is not appreciably dissipated by intermittent ventilation measures such as extract fans. However, the values seen are typically low for these cities and well below the 200Bq/m^3 action level (HPA, 2011).

In Milton Keynes, the housing stock is more recent and therefore built to a higher energy efficiency standard and greater airtightness. The housing typology also differs appreciably with over 50% of London's stock being purpose built flats (requiring simpler measures to obtain gains), while Milton Keynes stock comprises 80% detached, semi-detached and terraced dwellings with general larger building volumes (MKiO, 2014). This is reflected in the greater reduction in indoor sourced pollutants seen in the London stock, whilst the Milton Keynes stock generally having larger room volumes exhibited low concentrations for similar source emissions.

6.2.3. Health Impacts

The impact on health measured in terms of the *per capita* total of life years gained (Table 5.12) is greater in magnitude in London than in Milton Keynes. These impacts translate into increases in average life expectancy at birth of ~3 months (Milton Keynes) and ~4 months (London) with PPV, but *decreases* in life expectancy of ~2 months (Milton Keynes) and ~5 months (London) if PPV is not installed. This reflects the larger changes in the modelled indoor exposures in London, which are due to the greater potential for improving the housing stock primarily due to the greater age range of the London stock (generally older, less energy efficient dwellings). The results reveal that the inclusion of PPV has substantial bearing not only on the magnitude, but also the direction of health impact. Without PPV large increases occur in exposures to pollutants derived from indoor sources (Table 6.10), which more than offset the benefits of improved indoor heating season temperatures and protection against outdoor air pollution, resulting in substantial negative consequences for health overall in both settings.

Table 6.10 Modelled health impacts: changes in life years over 40 years for each scenario and per 1,000 population (brackets)

Scenario	Modelled change in life years over 40 years*	
	London	Milton Keynes
With purpose-provided ventilation		
Resilient and Low-Carbon Scenarios	849,800 (108.6)	21,200 (86.4)
Resilient + Scenario	856,500 (109.5)	21,400 (87.2)
Without purpose-provided ventilation		
Resilient and Low-Carbon Scenarios	-1,043,900 (-133.4)	-13,800 (-56.2)
Resilient + Scenario	-1,041,000 (-133.0)	-13,700 (-55.8)

6.3 Discussion

There are substantial differences in results for the two locations when housing interventions are the sole mechanism for decarbonization, without additional substantial grid decarbonization. London housing can achieve greater reductions in CO_2 emissions per capita (and possible average health net benefits) than that of Milton Keynes. This is as a result of various factors; the Milton Keynes housing stock is more recent and built to a higher energy efficiency standard. Type and distribution of housing differs appreciably with over 50% of London's stock being purpose built flats (requiring simpler measures to obtain gains), while Milton Keynes stock comprises 80% detached, semi-detached and terraced dwellings with generally larger building volumes that result in smaller concentrations of pollutants per unit volume (MKiO, 2014). To achieve similar reductions in CO_2 emissions in Milton Keynes would require a greater investment in more technical housing interventions such as mechanical ventilated heat recovery (MVHR) systems that were suitably maintained including servicing and the replacement of filters in a timely manner. The appropriateness of such interventions would of course require a detailed cost-benefit analysis. However, based on this study, it would appear that potential CO_2 reductions and health impacts (whether positive or negative) are stock-specific, being primarily influenced by building age- defining the building regulations they were built under, the distribution of archetypes, which effects room size and indoor pollutant concentration. It therefore follows that policies should be tailored to take this into account rather than be universally rolled out, with both regional and local strategies focusing on the most appropriate sectors in order to achieve CO_2 emissions reduction targets.

In addition, although a full sensitivity analysis has not yet been run on all the different components of the SCRIBE model, the uncertainties in the results seen in the last chapter are likely to be similar as the same baseline models were used. However, it is also possible that some of these may be reduced by the wider range of archetypes and variants. In both London and Milton Keynes, changes to the indoor environment following combined energy efficiency and PPV interventions (if perfectly implemented) would lead to lower CO_2 emissions, reductions in indoor $PM_{2.5}$ concentrations, and increases in indoor winter temperatures yielding average net health benefits. There is a distribution of values, some properties where PPV was not included as part of any refurbishment strategy would likely see increases in indoor pollutant concentrations and negative health consequences. In this respect our results are consistent with those of other published research. (Wilkinson et al., 2009; Crump, 2011; Milner et al., 2014). In scenarios where PPV (properly implemented) is used in conjunction with energy efficiency measures, the overall *per capita* health benefits (including all pollutant exposures and temperatures change) are greater in London, with benefits for cardiopulmonary health due to reductions in indoor exposure to both indoor and outdoor-generated $PM_{2.5}$. There would also be substantial reduction in lung cancer burdens due to the reduced $PM_{2.5}$ and radon levels with a minimal impact on ventilation heat loss in both locations. Providing PPV has impact on energy use of +8.6% on average between the different scenarios, whilst potentially yielding substantial health gains. However, a distribution of impacts will occur because of different housing geometries and occupant behaviours and for some homes and behaviours indoor exposures would increase and there would be health dis-benefits for some people. This could give rise to health inequalities (the subject of study for the next chapter) and needs further investigation beyond the scope of this thesis. Approved Document F1 of the building regulations states that following retrofitting ventilation should not become worse (HM Government, 2010), however on-site monitoring would have suggested this is not always the case (Sinnott & Dyer, 2012).

The UKERC carbon intensity scenarios used here assume the use of electricity as the energy source for space heating in the domestic sector, which is seen as essential under Low-Carbon scenarios for the UK energy system in 2050 because it can be generated from a range of renewable and low-carbon energy sources including nuclear and the use of carbon-capture technologies (UKERC, 2013). By combining housing interventions with decarbonization of the electric grid a substantial contribution to climate goals can be achieved, with targets exceeded in the UKERC Low-Carbon scenario in both locations. However, in both London and Milton Keynes domestic customer fuel consumption is currently 76% gas (DECC, 2014). It is likely that both legislative and incentive means will be needed to promote change from gas to an all-electric grid. If such a change is delayed or does not occur the predicted reductions in CO_2 emissions seen in this study will not be achieved. An energy efficient housing stock with (largely decarbonized) electricity as its fuel represents the upper limit of possible CO_2 savings.

6.4 Conclusions

This chapter has investigated the comparative impacts of dwelling-related CO_2 reduction strategies in London and Milton Keynes using integrated housing intervention and energy supply decarbonization scenarios to calculate possible end user energy demand, $PM_{2.5}$ (and other pollutant exposures) and health impacts. It has shown that variations in indoor $PM_{2.5}$ exist between locations. Where CO_2 reduction targets are the main policy driver, substantial reductions can be made in London with energy interventions on housing, whereas for Milton Keynes the potential percentage gains are much smaller because of the already more energy efficient housing stock confirming the work of Summerfield et al.

(2007). Potential net benefits or harms for health are also greater in London as measured in terms of *per capita* gains in life expectancy, highlighting the importance of not applying a 'one size fits all' energy saving and CO_2 emission reduction policy, as local differences in housing and the environment may have important bearing on the impacts that can be achieved. Decarbonization of the grid is essential in achieving CO_2 emissions reduction targets, especially in Milton Keynes. With the exception of Radon, which requires a separate mitigation strategy (explored in Milner et al, 2014 and chapter 8), overall reductions in indoor $PM_{2.5}$ and pollutant concentrations can be achieved by applying the same PPV strategies. However, when designing for both low energy use and good health, there are important trade-offs between an increase in the airtightness of dwellings and changes in IAQ. If interventions are not correctly applied, there are risks of serious negative health effects. In order to obtain both health gains and promote success in achieving CO_2 emission reduction targets in both locations, policymakers need to consider a wider view that includes strategies to extensively decarbonize the electricity grid with a move away from the reliance on residential use of gas.

Whilst this chapter has estimated relative differences between current and future indoor $PM_{2.5}$ (and other pollutant) exposure between two different types of building stock in England, the next chapter investigates a different aspect of $PM_{2.5}$ exposure; a possible social dimension to exposure in whether variations exist in $PM_{2.5}$ indoor exposure between different income groups and tenures.

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Chapter 7

Variations in Indoor $PM_{2.5}$ Exposure for Different Income Groups

Introduction

A report by the Joseph Rowntree Foundation (JRF, 2013) showed household CO_2 emissions are strongly correlated with household income. Low income households while consuming less energy and emitting less CO_2, use a higher proportion of their household income to pay for fuel, leading to possible social inequalities. However, whether this leads to variations in exposures to indoor pollutants such as $PM_{2.5}$ is currently unclear (JRF, 2013). Whilst the previous chapter considered variations in indoor exposure to a range of airborne pollutants - including $PM_{2.5}$ - between two cities, this chapter goes further and investigates possible differences in $PM_{2.5}$ exposure that may occur between dwellings of different tenures and occupant household income levels, as a consequence of the energy efficiency, built form, and ventilation characteristics of the houses. As previously shown, cooking is the most prevalent source of indoor generated $PM_{2.5}$ in non-smoking households. When coupled with energy efficiency and ventilation measures aiming to reduce overall CO_2 emissions, the impacts on $PM_{2.5}$ exposure for different income groups can vary. The type and quality of dwellings inhabited and the practices of the occupants may vary according to socio-economic status and income level, which may then further influence $PM_{2.5}$ exposure.

The UK Government, the European Union and many other countries define low-income households as those having a household income less than 60% of the national median income that year (DCLG, 2013). Occupants in houses below the lower income threshold (LIT) are more likely to live in smaller dwellings (occupant per m^2) such as flats, which may have lower air change rates then detached, semi-detached, or terraced dwellings due to the reduced number of external facades as seen in Taylor et al (2014a). Below LIT households may also differ from the overall building stock in terms of building retrofit levels. In addressing the socioeconomic and behavioural issues that influence the adoption of energy efficiency measures, Tovar (2012) concludes that households including single adults, those living alone or in cities, lone parents, and tenants in the private sector are the least likely to adopt cavity insulation, loft insulation, and boiler upgrades. Hamilton et al. (2014) however, showed that dwellings with the highest take-up of fabric interventions e.g. cavity wall insulation, loft insulation and glazing (the top 20%) are more likely to be found in areas with low income, in part attributable to council-led retrofits in public housing, and schemes such as Warm Front and the Energy Company Obligation (ECO) (Warm Front, 2004; ECO, 2014). These findings indicate a potential difference in building energy efficiency and airtightness between different income and tenure groups, which may have consequences for indoor air quality. Further investigation to clarify the possible impacts on health and to better inform housing policies aiming to target and improve energy efficiency of the housing stock is necessary

Occupancy and behavioural differences across income groups may also lead to differing levels of exposure to indoor air pollution. In the UK there is a strong link between smoking and income class, with 35% of unemployed adults smoking (compared to a rate of 19% in the economically active population). While smoking may not necessarily always occur inside the home, 59% of daily smokers surveyed allowed smoking in their homes adding to the $PM_{2.5}$ burden (ONS, 2007). This is likely to be elevated amongst those with mobility issues who are less able to leave their houses. In addition, extractor fans in poor housing may be more likely to remain unrepaired if broken or to underperform thereby reducing ventilation. Previous studies have demonstrated that in some UK cities (for example, London), below LIT income individuals live in areas of higher outdoor $PM_{2.5}$ than the general population (Pye et al, 2001; Tonne et al, 2008), while individuals of low socio-economic groups are the most susceptible to negative health consequences from pollution exposure (Deguen & Zmirou-Navier, 2010).

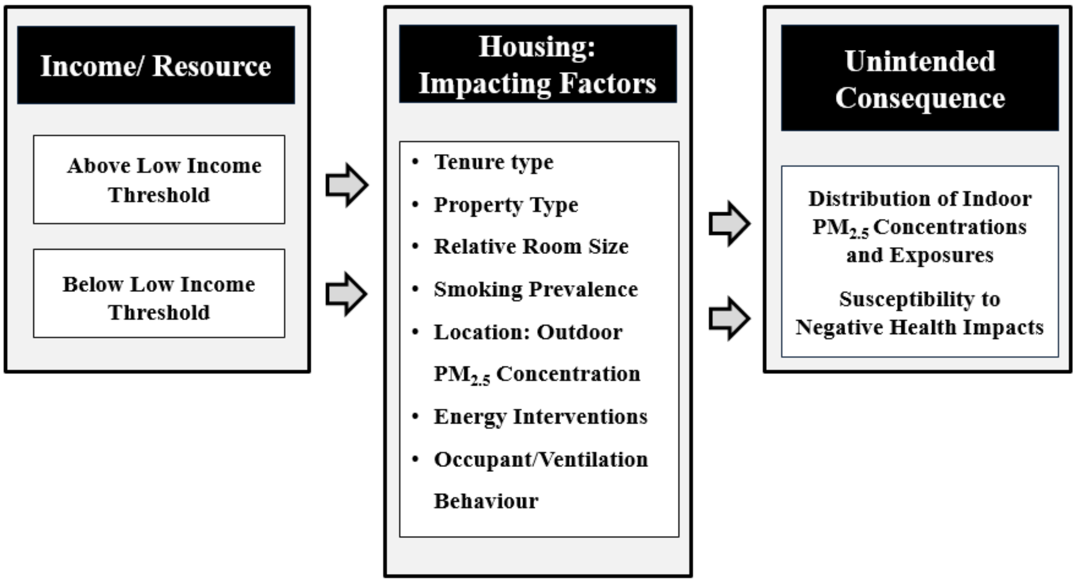

Figure 7.1 How income, tenure and behavioural factors affect $PM_{2.5}$ exposure.

This chapter examines how the existing English housing stock may modify the exposure to $PM_{2.5}$ from indoor and outdoor sources for below LIT households (and the various dwellings tenure groups within) and above LIT households, for both current and full levels of retrofit as seen in the previous chapter. Using EnergyPlus, an energy analysis and thermal load simulation program with a multizone airflow and contaminant transport analysis component (US-DOE, 2013), simulations were run for the infiltration of outdoor $PM_{2.5}$ into the indoor environment and indoor sourced $PM_{2.5}$. Subsequent to the previous chapter, the Generic Contaminant Model (GCM) in EnergyPlus was introduced which allows for the integrated modelling of multizone contaminant and dynamic thermal behaviour within a single simulation package. Prior to this, pollutants could not be modelled in EnergyPlus, whereas dynamic thermal profiles had to be imported into CONTAM from EnergyPlus. The importance of taking into account dynamic thermal effects in simulations of pollution transport in buildings and programme comparisons are dealt with in Taylor et al (2014c), to which the author contributed. Using this programme all the modelling components and assumptions previously in CONTAM were duplicated which enabled faster processing of results and was therefore chosen for use in this study. In addition, this 'new' set of models were used in other projects beyond the scope of this thesis.

Simulations generated include a set of models representing the range of ages and built forms in the current English housing stock and possible fully retrofitted stocks under the different tenancies. The results for each model were weighted according to the frequency of occurrence for each age and built form combination in the different groups (above and below LIT) in order to calculate the differences in total $PM_{2.5}$ exposure between them. Finally, a series of statistical tests were carried out to test for differences between the different income and tenure groups.

The investigation in this chapter resulted in a peer reviewed research paper (Shrubsole et al., 2015) as well as a successful application for funding from the Public Health England (PHE) PhD Studentship Fund, proposed to be supervised by PHE, UCL and University of York, including the author as a subsidiary supervisor. The title of the bid is: 'Quantifying the benefits of measures to reduce exposure of deprived communities to indoor and outdoor sources of air pollutants'. The integrated modelling tool to be used in this PhD is based around the modelling seen in Chapter 5 (Shrubsole et al., 2012) and the issues raised in this chapter (Shrubsole et al., 2015).

7.1 Methods

The basis of this study was the 2010-2011 English Housing Survey, a statistically representative survey, which contains information on housing characteristics and the occupant households for around 16,000 dwellings. Household survey data includes information on income level and tenure, while physical survey data contains physical data on the dwellings themselves. Each entry in the EHS is associated with a dwelling and household weight, representing the dwelling or household occurrence within England, in addition to a wide range of data describing dwelling characteristics and their inhabitants.

7.1.1 Development of Representative Archetypes

In this chapter, a set of 11 archetypes representing the range of built forms in the EHS characteristic of the whole English housing stock (Figure 7.2) were constructed with multiple variants having various energy efficiency and ventilation interventions added -see chapter 4 and table 4.2 for further details. These used archetypes of dwellings from Oikonomou et al (2012) and the AWESOME project (2013) and are assumed to broadly represent the English domestic stock. The expanded set of archetypes took advantage of the newly-developed AWESOME archetypes to supplement existing London archetypes. Where there were built forms (geometries) with multiple archetypes (e.g. terraced dwellings: mid or end), the simulation results were averaged across the variants to determine a single value for the built form. The resultant eight built forms matched the categories for building type in the EHS.

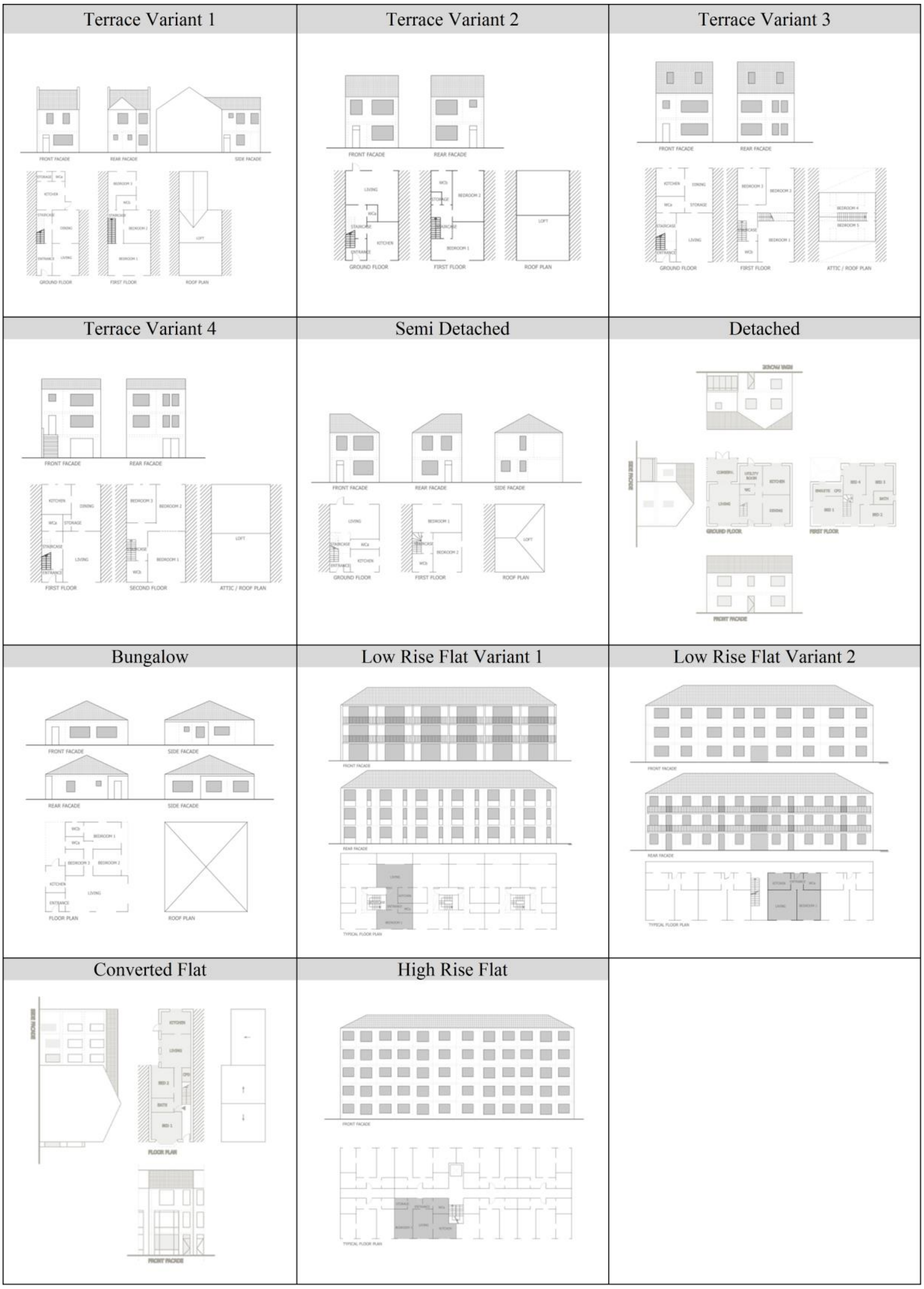

Figure 7.2 Representative archetypes used to investigate the EHS data base

7.1.2 Dwelling Ventilation

The building fabric permeability of individual dwellings in the EHS was estimated using the UK Standard Assessment Procedure (SAP) methodology (BRE, 2009) as per DECC (2012a) with corrections for sheltering applied as seen in Taylor et al (2014a), with the exception that draught proofing and floor sealing were excluded from the calculation, as their influence on permeability was to be considered separately.

Instead of static opening schedules for windows based around time of year as seen in the CONTAM modelling in chapters 5 and 6, as EnergyPlus enables dynamic temperature modelling, dwelling window-opening behaviour was coupled to indoor temperatures, as carried out in Taylor et al. (2014a). Living room windows were considered to be opened during the day if the internal temperatures exceeded 25°C, while bedroom windows were considered to be opened during the night if temperatures exceeded 23°C. In both cases, windows remained closed if the indoor temperatures were less than that outdoors. While there are a number of factors which may influence occupant window-opening behaviour, internal temperature is one of the most significant, and the thresholds used in this study are in line with those observed in field studies (Dubrul,1998; Fabi et al., 2012) and CIBSE overheating guidelines (CIBSE, 2006).

Purpose-provided ventilation was modelled using extract fans in the kitchen and bathroom. These fans were modelled with a flow rate of 60l/s based on the fan not being adjacent to the hob, as per building regulations (ADF 2010), and operated during cooking. Analysis of data from the English Housing Survey (EHS) variables found that there was a slight difference in levels of working kitchen extract fans (EHS parameter *finkxtwk*) across the income and tenure groups with below LIT income households working 44.5% of the time they should be in use, above LIT income households 48.4% of the time.

7.1.3. Occupant Behaviour

A single occupancy scenario representative of a family was modelled. The family was assumed to be absent from the dwelling during weekdays between 9am to 5pm, and home all day during the weekends. Dwellings were assumed to be heated to 20°C during the night throughout the year, while internal gains from electrical equipment and occupant metabolism were also included in the model as seen in Taylor et al. (2014b) in order to ensure all heat sources were accounted for.

LIT groups are defined as having below 60% of the median income for that year, weighted by peoplegross (*AHCinceqv60h*) parameter in the EHS). In below LIT dwellings, 44% were found to have at least one occupant that smoked, with similar levels across tenure groups, while 28% of above LIT dwellings were found to have at least one occupant that smoked. Household smoking data in the EHS was used to determine the presence of at least one smoker in each EHS variant, as specified by the *cignow* parameter in the EHS.

$PM_{2.5}$ levels and emission schedules were modelled as per the studies seem in the schedule of activities for chapters 5 and 6 and can be seen in full in Table 7.1 while the $PM_{2.5}$ emission rates, outdoor particle penetration factor, and deposition rates can be seen in Table 7.2.

A different deposition rate was considered for Environmental Tobacco Smoke (ETS) due to the different size fraction of $PM_{2.5}$ that characterises the majority of ETS. Two ventilation scenarios were modelled during cooking with the extractor fans either on or off, while no additional ventilation was used when smoking occurred indoors. Although it is likely that the different constituents of $PM_{2.5}$ pose different risks to health, given the lack of evidence in this area, we have assumed that from $PM_{2.5}$ indoor sources are equally as toxic as those found in outdoor air.

Table 7.1 Indoor $PM_{2.5}$ production schedules.

Activity	Location	Schedule
Cooking	Kitchen	07:45 – 08:00 12:00 –12:30* 19:00 - 19:30
Smoking	Kitchen	8:00 – 8:05 9:00 – 9:05
	Living Room	10:00 – 10:05* 11:00 – 11:05* 12:00 – 12:05* 19:00 – 19:05 20:00 – 20:05 21:00 – 21:05 22:00 – 22:05

*represents those events that only occur on weekends.

Table 7.2 $PM_{2.5}$ emission rates, outdoor particle penetration factor, and deposition rates

Source	Penetration Factor	Annual Outdoor Level	Emission Rate	Deposition Rate
Outdoor	0.8 when windows closed[1] 1.0 when windows opened[1]	13μg/m^2[3]	_	0.19h^{-1}[3]
Cooking	_	_	1.6mg/min [4]	0.19h-1[5]
Smoking	_	_	0.9mg/min [4]	0.10h-1[6]

[1, 3 & 5]Long et al., 2001; [2]Shrubsole et al., 2012; [4]Dimitroulopoulou et al., 2006; [6] Klepeis & Nazaroff, 2006

7.1.4. Retrofits

Using the EHS data, Figure 7.4 shows the current levels of retrofit across the various tenure categories within the below LIT group, and for the above LIT income group. Measures include loft, floor and wall insulation and double glazing in this scenario. Below LIT private-rented dwellings tend to have the lowest levels of retrofit reflecting the lack of decision making autonomy for either accepting or seeking energy efficiency improvements. The owner-occupied below LIT and above LIT-income categories have the second lowest levels of retrofit. The below LIT local-authority and registered social landlord (RSL) housing tend to have the highest levels of retrofit (Hamilton et al., 2014).

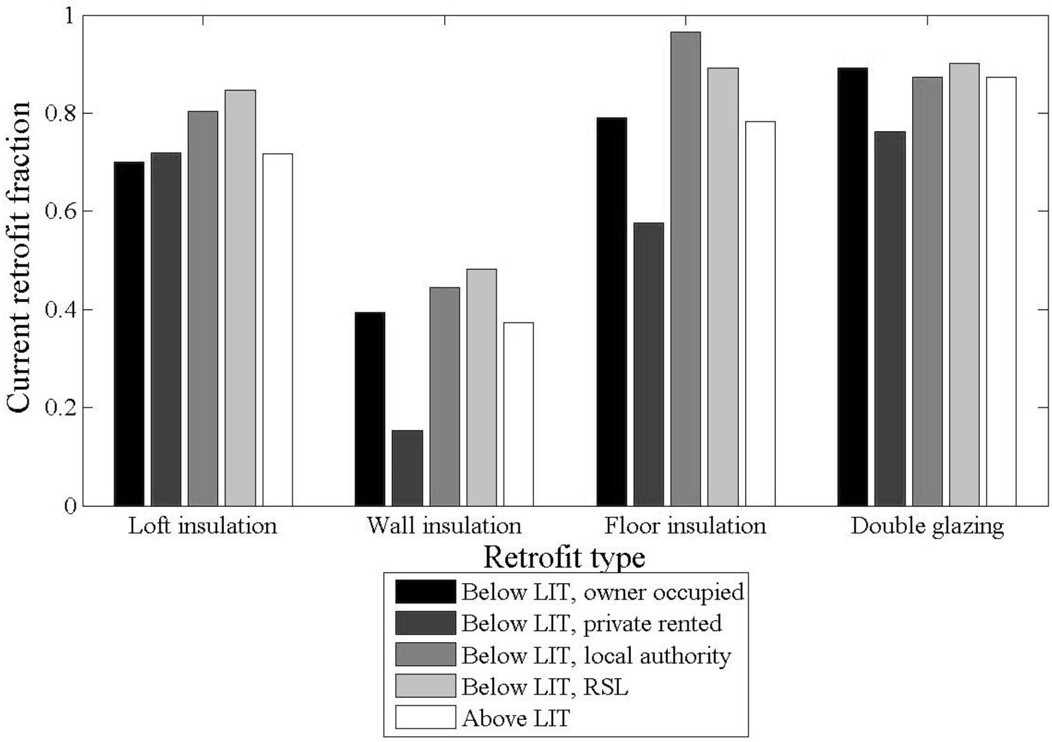

Figure 7.3 Current levels of various retrofit measures across income and tenure groups (source: data from 2010 EHS, graph produced by author).

The four potential retrofit measures examined included wall and loft insulation, floor sealing, and double-glazed windows (used as a proxy for draught-proofing). These retrofits were selected as they are thought to be some of the largest contributors to infiltration according to the Warm Front study (Hong et al, 2004).

Changes to dwelling permeabilities caused by wall, loft, floor, and window retrofits were calculated based on estimates from the Warm Front study (Warm Front, 2004) (Table1). The current levels of retrofit were estimated for each dwelling in the EHS, based on the presence of variables reflecting wall, window, and loft improvements. Individual retrofitting measures were applied to all those properties not currently having them. All pre-1919 dwelling were assumed to have suspended floors and be therefore eligible for floor retrofits (i.e. the sealing or concreting of a suspended floor). The presence of individual retrofits interventions was used to adjust the SAP-calculated permeability accordingly. Additionally, an estimate of the final building permeability following implementation of all four types of retrofit was calculated. It was assumed that retrofits were carried out without any additional compensatory ventilation (a worst-case scenario), and that building permeability did not drop below $3m^3/hr/m^2$@50Pa. Figure 7.4 shows the process of obtaining building envelope permeabilities for buildings post retrofit.

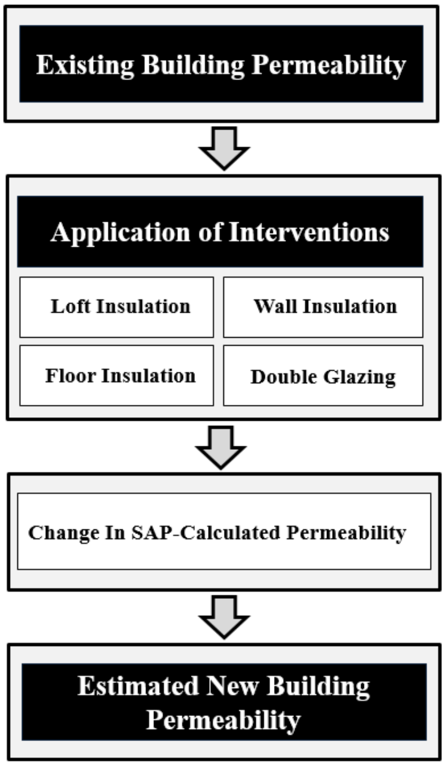

Figure 7.4 Process of obtaining post-retrofit building permeabilities.

Table 7.3 Percentage change in permeability following retrofits.

Retrofit measure	% Change in permeability
Pre-retrofit (PR)	0
Wall Insulation (WR)	-9
Loft Insulation (LR)	-14
Floor Sealing (FR)	-17
Double-Glazing/Draught Proofing (DGR)	-5

7.1.5 Dynamic Building Simulation

Simulations were constructed and run in EnergyPlus 8.0. EnergyPlus was used for the study in this chapter rather than CONTAM, as the introduction of the Generic Contaminant Model (GCM) enabled the temperature-coupled modelling of a single pollutant. Coupled temperature-pollution modelling was not previously possible in CONTAM without running thermal simulations in a secondary program and importing the results as a schedule into CONTAM. Furthermore, developing the models in EnergyPlus allowed for the integration of indoor temperature and pollution modelling frameworks developed at UCL. As discussed in Chapter 3, (section 3.4) the EnergyPlus GCM and CONTAM operate using the same underlying modelling principles and have been demonstrated to give similar results (Taylor et al., 2014c). Simulations were run for an entire year with both outdoor and indoor sources of $PM_{2.5}$ (smoking, cooking, and cooking without ventilation). The EnergyPlus (EP) base line archetype variants comprised each of the built forms modelled at eight different permeability levels (3, 5, 7, 10, 15, 20, 25, and 30

$m^3/hr/m^2$@50Pa), with the more airtight dwellings (3, 5, and 7 $m^3/hr/m^2$@50Pa) modelled with fabric characteristics with greater thermal insulation levels. This covered the full range of characteristics of the current and possible fully retrofitted housing stocks under different levels of retrofit. Each EP variant was also modelled assuming four different orientations (North, East, South, and West), to enable orientation-averaged outputs to be evaluated, and both with and without trickle vents. Weather conditions were modelled using a Typical Reference Year (TRY) weather file for Central London (Islington) obtained from the Prometheus project (Eames et al., 2011) and considered sufficiently indicative of general urban conditions in England for the purposes of this study.

Contaminant modelling in EP was performed using the Generic Contaminant Model tool. Models were run with a constant outdoor background concentration for $PM_{2.5}$ of 13µg/m^3. The Generic Contaminant Model requires contaminants be modelled generalised to a volume of air with mass equivalent to the pollutant mass. Consequently, contaminant emission rates from indoor sources were converted to m^3/s using the density of air at 20°C.

7.2 Data Output and Analysis

7.2.1 Data Collation and Matching

The hourly pollutant concentrations in the living room, bedroom, and kitchen were output from the simulations as representing those rooms most frequently occupied. The EP output files were collated and analysed using the SAS statistical package (SAS, 2013), and used to calculate the pollutant concentrations occupants were exposed to, based on the room occupied at the corresponding schedule time. The annual average concentration of $PM_{2.5}$ from outdoor sources, cooking, cooking without extract fans, and smoking (in absolute concentration, µg/m^3) were averaged across the four building orientations for each simulated EP built form/permeability variant. $PM_{2.5}$ from indoor sources were converted from ppm back to µg/m^3 using the density of air at 20°C.

Indoor pollution levels from different sources were then assigned to each entry in the EHS based on the built form and estimated current and complete-retrofit permeability by interpolating between the different modelled permeability levels. As there is no parameter indicating the presence of trickle vents in EHS dwellings, results were weighted for assumed trickle vent presence for pre and post-1990 dwellings, based on Warm Front (DECC, 2011) observations that suggested that 10% of pre-2002 dwellings have trickle vents and 100% of post-2002 dwellings. As the latest age bracket in the EHS is post-1990, we assumed a constant annual construction rate, and adjusted the rates of new builds accordingly. The presence of a working extractor fan in the kitchen in each EHS entry was used to indicate whether indoor pollution levels from cooking were with or without such a ventilation system. Smoking was similarly weighted: if a smoker was not present in the EHS variant for smoking, the $PM_{2.5}$

concentration from smoking was assumed to be zero. Estimates of the current variation and likely changes in $PM_{2.5}$ exposures following a full retrofit of the housing stock across tenure and income categories were then examined.

7.3 Results

The mean indoor $PM_{2.5}$ concentrations for the current housing stock and a fully retrofitted housing stock derived from the EP simulations are shown in Figure 7.5. These include both $PM_{2.5}$ from outdoor sources and indoor sources including smoking and cooking across the various income and tenure groups.

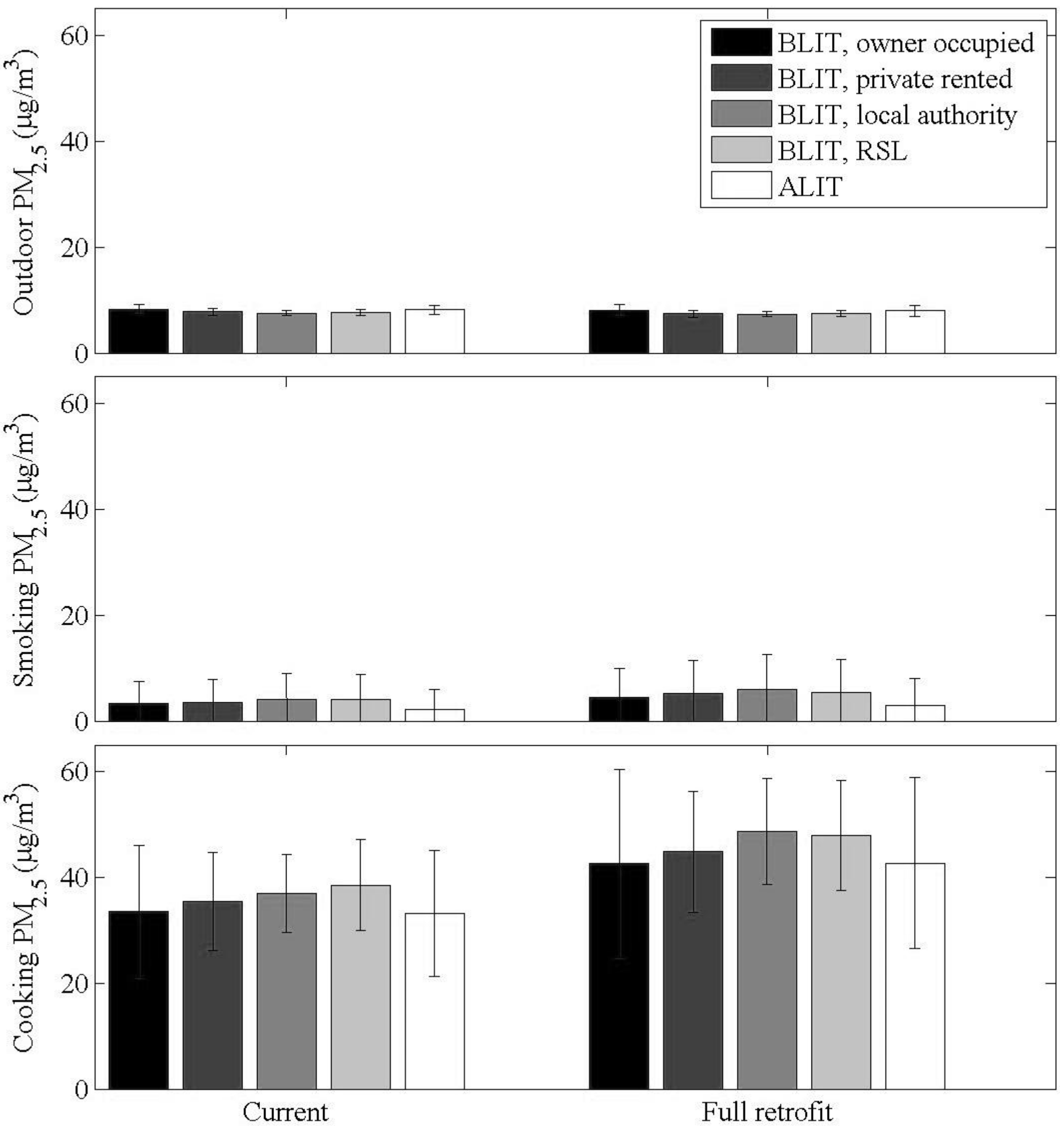

Figure 7.5 Indoor $PM_{2.5}$ exposure indoors from different sources for current and fully retrofitted scenarios, and across income and tenure groups. The error bars show standard deviations, and are large for smoking due to some dwellings having zero concentrations. The cooking $PM_{2.5}$ is the exposure

experienced by cooks in the kitchen of the properties (BLIT= below low-income threshold, ALIT= above low-income threshold). From this it can be inferred that those who undertake the majority of the household cooking may experience greater levels of exposure compared to non-cooks, whilst they are both exposed to similar levels of externally generated $PM_{2.5}$. There is also a suggestion that below LIT-income groups are at higher risk of exposure to greater concentrations of $PM_{2.5}$ when compared to above LIT-income groups due to smaller houses and a smaller number of exposed facades leading to a reduced air change rate. In addition, they may experience higher rates of smoking and greater likelihood of cooking without working extractor fans compared to ALIT households. It appears that the fully retrofitted housing stock poses a higher health risk compared to the current housing stock primarily due to a general reduction in building permeability and consequent air change rate following retrofitting interventions on the building envelope, confirming previous conclusions in chapters 5 and 6. Whilst this has the effect of reducing the ingress of outdoor sourced $PM_{2.5}$ it results in an increase in concentrations of indoor sourced $PM_{2.5.}$ The simulations show that cooking is clearly the biggest contributor of $PM_{2.5}$ to the indoor environment in smoking and non-smoking households, as previously shown and that cooks therefore receive greater exposures than occupants not present in the kitchen. However, it is noted that the relative levels of $PM_{2.5}$ from smoking sources are less than those seen in the studies in chapters 5 and 6. There may be a number of possible causes that could explanation this reduction. The EHS has a flag for houses with a working extract fan, which are different to a particular stock such as the GLA or Milton Keynes. Consequently, calculated values were at the stock level. This therefore means that all houses had cooking, almost half of which didn’t use an extract fan. Only a fraction of these had smoking as a source.

A series of one-way ANOVA tests were carried out in MATLAB (MathWorks, 2012) to further clarify the results and test for differences between the income and tenure groups within each of the current and fully retrofitted housing stocks, and between the current and fully retrofitted housing stocks as a whole. As there are more than two groups when comparing between income and tenure groups, MATLAB’s multiple-comparison tests were subsequently carried out if the initial ANOVA test found a significant difference. These isolated the location of the differences whilst ensuring Type-II errors were adequately accounted for. The results are shown in Table 7.4

Table 7.4 Results of ANOVA tests for the difference $PM_{2.5}$ sources. 'Yes' signifies a difference at the 95% level of confidence, $p>0.05$. The 'details' column summarises the differences as derived from the multiple-comparison tests.

Pollutant	Between current income/ tenure groups	Between retrofitted income/ tenure groups	Between current/ retrofitted groups	Details
Outdoor $PM_{2.5}$	Yes	Yes	Yes	Below LIT-income, owner-occupied and above LIT-income groups are similar to each other but different from other groups in current housing stock, though the groups are more similar in the fully retrofitted housing stock.
Smoking $PM_{2.5}$	Yes	Yes	Yes	Above LIT-income group is different from all other groups in both current and fully retrofitted housing stocks.
Cooking $PM_{2.5}$	Yes	Yes	Yes	Below LIT-income, owner-occupied and above LIT-income groups are similar to each other, but all the other groups are significantly different from these and from each other in the current housing stock. Similar for fully retrofitted housing stock but below LIT-income local-authority and RSL groups more similar to each other.

The ANOVA tests support significant differences in all cases at the 95% level of confidence, although it is acknowledged that this reflects differences between the modelled $PM_{2.5}$ exposures rather than actual exposures. Actual exposures may exhibit different distributions as a result of uncertainties in model variables such as the behaviour of occupants, which may also vary across income and tenure groups; dwelling characteristics that are not informed by the described data sources and variations in weather variables across dwelling locations. In the case of comparing modelled exposures between current and fully retrofitted housing stocks, the ANOVA tests highlight significant differences in the concentrations of different sources of $PM_{2.5}$ indoors: outdoor $PM_{2.5}$ decreases, smoking $PM_{2.5}$ increases, and cooking $PM_{2.5}$ increases.

7.4 Discussion

This study provides new insights into the predicted average relative differences in indoor $PM_{2.5}$ exposure that exist between the various income and tenure groups of the English domestic stock. These differences in exposure are primarily driven by differences in the dwelling characteristics they occupy, but also their habits, such as smoking. In addition, the study describes the potential impacts of changes to occupant $PM_{2.5}$ exposure following an energy efficiency retrofitting scenario. Results generated from the computer modelling were analysed further to determine the statistical limits of the relative differences using ANOVA tests. Exposure levels modelled are generally consistent with previous research using different modelling programmes (chapters 5 and 6) and techniques (e.g. Milner et al., 2005; Shrubsole et al., 2012; Gens et al., 2014), which shows that the application of energy efficiency interventions on the domestic stock, whilst reducing exposure to outdoor sourced $PM_{2.5}$ may increase exposure to indoor sources. However, it is acknowledged that these are each different scenarios, having different criteria and focus and are therefore not directly comparable. It is also acknowledged that the choice of occupant schedules and related activities impact the indoor $PM_{2.5}$ exposure. Future work could develop a full range of schedules for different household types and explore occupant behaviour in greater detail.

It would appear that below LIT income groups have, on average, higher levels of exposure to $PM_{2.5}$ across the building stock when compared to above LIT income groups. This may in part be due to the greater uptake of measures that reduce the permeability of the building envelope particularly by councils and housing associations and therefore lower air change rates where additional purpose provided ventilation is not provided or maintained. However, it is acknowledged that within each income band there will be a range of individual personal indoor $PM_{2.5}$ exposures. Furthermore, as with all modelling studies, a number of assumptions are required, and further empirical investigation is necessary to confirm or refute the findings. The primary $PM_{2.5}$ source appears to be from cooking, and therefore the provision, use, and appropriate maintenance of adequate extraction equipment (e.g. cooker hoods) is essential to remove this pollutant. This could reduce the apparent increase in $PM_{2.5}$ concentrations and still keep the benefits of increased insulation such as greater thermal efficiency. Assistance with fuel costs whilst encouraging better ventilation behaviour may also increase relative CO_2 emissions undermining reduction policies.

Comparisons between groups in each housing stock using the ANOVA multiple-comparison tests show that below LIT-income owner-occupied and above LIT-income groups have higher levels of outdoor $PM_{2.5}$ in the current housing stock, although the differences are small and most likely due to the lower levels of retrofit shown in Figure 7.2. However, these differences are not seen in the housing stock, following full retrofit. The above LIT-income groups have lower levels of $PM_{2.5}$ from smoking in both the current and fully retrofitted housing stocks compared to all other groups, primarily as a result of a lower number of households with occupants who smoke rather than other factors. $PM_{2.5}$ sourced from cooking is lower in above LIT-income dwellings in both the current and fully retrofitted housing stocks, and is also lower in owner-occupied and private-rented below LIT-income dwellings compared to local-authority and RSL below LIT-income dwellings. These may be a result of higher levels of retrofit in the local-authority and RSL below LIT-income dwellings, but as these differences persist in the fully

retrofitted housing stock, it may also be a result of other factors, possibly generally smaller dwelling/kitchen sizes.
Although in all airtightening scenarios the ingress of outdoor $PM_{2.5}$ is seen to reduce, as has been previously noted, below LIT income individuals tend to live in areas of higher outdoor $PM_{2.5}$ than the general population (Pye et al, 2001). This may act to counter the advantage of the below LIT-income social housing, which were found to have lower levels of indoor $PM_{2.5}$ from outdoor sources, while further increasing the risks to below LIT-income individuals in privately-rented accommodation.

The use of window opening to ventilate dwellings and thereby improve IAQ has been found to be less likely amongst elderly occupants, possibly due to a preference for higher indoor temperatures (Dubrul, 1988; Guerra-Santin et al., 2009). No significant correlation was found between socioeconomic factors and window opening behaviour (DuBrul, 1998). However, it is reasonable to assume that in poorer areas where there is either fear of or actual criminal activity, occupants may be less likely to leave their windows open for security reasons (Fabi et al., 2012), which may further increase disparity in exposure to pollution from indoor sources between income groups. Other factors influencing indoor domestic $PM_{2.5}$ exposure in below LIT income dwellings that require further investigation are the possibility of overcrowding, and multiple smoking occupants which are known to be more prevalent in below LIT income dwelling and add to the $PM_{2.5}$ exposure risk. In addition, the reductions in permeability which decrease air change rates may encourage the transmission of airborne infections and diseases in below LIT income properties (Beggs et al., 2003; Noakes et al., 2006). This is particularly relevant to the private rented sector which is currently growing and is less regulated when compared to Local Authority or RSL dwellings.

It is acknowledged that whilst $PM_{2.5}$ has known negative health impacts, there are other indoor airborne pollutants e.g. volatile organic compounds (VOC), radon and mould which each have associated health effects (Wilkinson et al., 2009, Milner et al., 2014). The trade-off that exists between airtightness and the consequent reduction of ventilation heat loss to achieve GHG reduction goals and public health concerns for IAQ have been previously noted (Wilkinson et al., 2009; Davies & Oreszczyn, 2012). Consequently, an inclusive optimum strategy approach is needed for building ventilation (Jones et al., 2013b) if health is to be a key driver of policy rather than a singular focus on decarbonisation (Crump et al., 2011).

This trade-off between the need for adequate ventilation to improve IAQ, comfort and energy conservation on a limited budget may also add to personal $PM_{2.5}$ exposure profiles for below LIT income occupants. Airtightening in order to conserve energy will likely also have the effect of raising indoor temperatures during summer months (Mavrogianni et al., 2012). This may lead to changes in occupant ventilation behaviour influencing IAQ. This dilemma has been successfully investigated in our study by using coupled thermal/pollutant modelling that is able to account for the increase in outdoor sourced $PM_{2.5}$ found indoors when occupants ventilate their properties when temperatures become uncomfortably high in the summer. It has also been noted that $PM_{2.5}$ external levels are generally lower

in the summer mainly due to metrological impacts, primarily convection and dispersal (McMurry et al., 2004) thereby lessening this effect, however this may not be the case for all pollutant types. In addition, the lower I/O ratio seen in below LIT income housing may also be offset by the location of many such properties in areas with generally higher outdoor pollution levels as previously noted.
Future research could investigate tenure in more detail for properties above LIT, subject to the availability of data, which could help more fully disentangle the differences between tenure and income. The value of this modelling approach is to help understand the main factors which impact on the complex relationship between building characteristics, occupant behaviour and indoor/outdoor sources, all of which could be partly determined by income/tenure/behaviour in conjunction with various retrofit measures Further research could also consider further sensitivity analysis or a regression modelling approach to examine the relative contribution of some of these aspects on the outcome (i.e. indoor PM2.5 exposure).

7.5 Conclusion

This study has developed and applied a series of stock model simulations in EP in order to quantify the changes in indoor domestic $PM_{2.5}$ exposure within the English housing stock that occur when buildings are retrofitted with energy efficiency measures. These results have been further subjected to a rigorous statistical analysis to confirm trends of the differences in model estimates. This study highlights the possible unintended consequence of changes to indoor domestic $PM_{2.5}$ exposures occurring as a result of retrofitting measures and the health trade-offs that may occur when policies to mitigate climate change do not take into account wider health outcomes. Results indicate that, on average, all types of low income households below the LIT experience greater overall concentrations of $PM_{2.5}$ than those above the LIT and suggest possible social inequalities driven by housing, leading to consequences for health. Below LIT income properties and especially those which are not owner-occupied are generally shown to be more vulnerable to increased levels of indoor $PM_{2.5}$ from indoor sources when compared to above LIT income properties, with $PM_{2.5}$ from cooking being the main cause. The increased use of extraction equipment at source could remedy this. Below LIT income housing represents a complex situation with multiple factors - physical, social and economic -influencing occupant exposure to pollutants such as $PM_{2.5}$. Whilst tightening the building envelope to save energy and assist with climate change mitigation objectives is laudable, it is essential that adequate ventilation is provided to avoid the negative health impacts.

The modelling work in this chapter has shown that there are potential social inequalities arising as a result of the application of energy efficiency interventions and that these impact on current and future $PM_{2.5}$ indoor domestic exposure for different social economic groups. Although it is acknowledged that further empirical investigation is required to confirm/refute these findings, they do offer insights into the impact of current policies.

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Chapter 8

Summary Discussion, Conclusions and Future Work

Introduction

The purpose of this chapter is to articulate a summary of the main evidence arising from the thesis in relation to the research objectives; the limitations of the study and how these impact the results and their interpretation; conclusions drawn, as well as implications for policy, practice and further research.

This research addresses the question as to whether the introduction of climate change mitigation strategies - policies that encourage the application of energy efficiency and ventilation measures - will lead to negative unintended consequences by increasing $PM_{2.5}$ concentrations in English dwellings, or provide co-benefits for health by a reduction of $PM_{2.5}$ exposure. Whether occupant behaviour, location, income and tenancy are factors that have a modifying influence on personal exposure, how all these issues could be balanced in order to consider an optimal strategy. Additionally, that all these factors can be examined using validated computer modelling software. In order to investigate these issues a number of research objectives were established as noted in Table 1.1 section 1.3. These have all been achieved as elaborated upon in Table 8.1 below.

8.1 Summary of Evidence

Having established the aims and research objectives of this study, chapter 2 provides the context for the study by investigating the impacts of policy drivers, whose main aim is to reduce CO_2 emissions in the UK domestic stock. This primarily single-focused policy has led to a range of '*unintended consequences'*, of which changes in indoor air quality (IAQ) and in particular concentrations of $PM_{2.5}$ is an important impact due to the known negative health impacts of $PM_{2.5}$ exposure. This chapter also confirms that many of the impacts affecting $PM_{2.5}$ exposures and emissions strategies are connected in complex and dynamic relationships, and that a possible way forward in order to explore these issues in terms of future policy construction is to adopt a more integrated approach to decision making which could ensure co-benefits are optimised, negative impacts reduced and trade-offs are dealt with explicitly. Such integrative methods -for example participatory system dynamics (PSD)- can capture this complexity in a policy making context and support a dynamic understanding of the effects of such policies over time. Such cross-sectorial policy optimisation places reducing housing GHG emissions alongside other housing policy goals and could assist in such areas as changes in IAQ and more specifically indoor concentrations of $PM_{2.5}$. It should be stressed that as $PM_{2.5}$ is one of many indoor airborne pollutants, optimal strategies should be considered which take into account all pollutants present in the indoor environment as well as impacts on indoor temperature.

From this perspective, a description of the characteristics of $PM_{2.5}$; the resulting health impacts from exposure; the many factors influencing indoor concentrations of $PM_{2.5}$ and methods of measurement are discussed in detail. This study material shaped the research methodology seen later. A range of methods for investigation was considered and a modelling approach was adopted using validated software, techniques and procedures, in part due to the levels of complexity of the investigation, the cost of monitoring and the need to explore future scenarios in order to research the possible future variation in indoor $PM_{2.5}$ concentrations and the consequences of the application of energy efficiency and ventilation intervention on the English domestic stock. In addition, the process for selection of the various inputs required in order to investigate the questions posed by this thesis were detailed.

In an idealised case study, the investigation into London's domestic stock in chapter 5 using two archetypes and their variants provided new insights into the potential effect of changes to the energy efficiency of London's housing stock on exposure to $PM_{2.5}$ by introducing MVHRs into substantially air-tightened homes. The results suggest that domestic energy efficiency interventions of the type and scale needed to meet 2050 climate change abatement objectives could yield substantial net reductions in $PM_{2.5}$ exposures. However, this is very much an idealised situation and assumes fixed occupant behaviour, correct installation and running of MVHR equipment with ventilation rates in line with the requirements of the Building regulations (ADF, 2010), none of which are by any means certain. These potential reductions in $PM_{2.5}$ exposures could lead to potential health gains, but such exposures varied according to the activities/behaviour of the occupants. In the existing (2010) stock, the outputs quantified via modelling showed the that average whole-house exposures did not necessarily indicate individual exposure and that in non-smoking homes the cook was subjected to twice the whole-house exposure. Such exposures may be addressed in part by the use of cooker hoods/extract fans at the point of source if run during and after cooking episodes. However, the current building regulation specifically excluded smoking as a basis for proposed ventilation rates. Comparisons with both monitoring and other modelling studies, confirmed the effective use of the modelling techniques for pollution investigation with outputs showing that results for the 2010 stock are broadly consistent with on-site measurements of average annual indoor domestic $PM_{2.5}$ concentrations and with those from modelling. With the use of MVHR systems, appreciable reductions in $PM_{2.5}$ exposure are also seen in smoking dwellings, thereby also reducing the health impacts for non-smoker in smoking households. Modelling with OSPM showed that there was some variation in external $PM_{2.5}$ concentrations entering the buildings, however in the particular scenario here, as the buildings were made very airtight and MVHRs filtered 80% of the external $PM_{2.5}$ this had little impact. Sensitivity analysis on model variables showed the range and distribution of concentrations in the London stock and also $PM_{2.5}$ emission and deposition rates with window opening behaviour to be the most effective variables. However, there was a difference in the strength of evidence for some variables. Consequently, results should be treated as indicative.

In a more realistic case study in chapter 6, a broader variety of building interventions were explored in order to investigate in greater depth the distribution of $PM_{2.5}$ concentrations alongside the impacts of a variety of interventions plusa broader range of archetypes (linked to the English Housing Survey) more indicative of the whole London and Milton Keyes domestic stock. Its function is to compare geographical locations to see if the capability to achieve CO_2 reductions (the policy context) were possible and whether the impacts in indoor $PM_{2.5}$ concentrations varied between locations. Results show there are substantial differences between the locations, primarily as Milton Keynes is a more recent stock subject to more stringent building regulations, which makes further energy efficiency gains harder to achieve. However, in both locations reductions in indoor $PM_{2.5}$ concentrations are seen where purpose provided ventilation (PPV) is incorporated as part of any energy efficiency intervention leading to health benefits from a potential reduction in airborne pollution. Conversely, where PPV is not used in a scheme, buildings are air-tightened to such a degree that the increase in indoor concentrations are likely to lead to CO_2 reductions from end user energy demand being achieved at the price of negative impacts on human health due to increases in indoor $PM_{2.5}$ (and other pollutants). This appears to outweigh health gains due to increases in indoor temperatures, which would reduce cold weather mortality. This chapter re-emphasises the trade-off that can exist between emission reduction goals and health benefits. By investigating other pollutants alongside $PM_{2.5}$ and indoor temperature, the need for optimal strategies is highlighted.

In chapter 7, which considered the whole English housing stock, results indicate that, on average, all types of low income households below the Low income threshold (LIT) experience greater overall concentrations of $PM_{2.5}$ than those above the LIT and suggest possible social inequalities driven by housing, leading to consequences for health. Below LIT income properties and especially those which are not owner-occupied are generally shown to be more vulnerable to increased levels of indoor $PM_{2.5}$ from indoor sources when compared to above LIT income properties, with $PM_{2.5}$ from cooking being the primary cause.

Within the above chapters 5-7, there are slight variations seen in predicted indoor exposure levels for $PM_{2.5}$ for the current day (2010). In the case of chapters 5 and 6 this is due to a greater range of archetypes and variants being used to model the stock in chapter 6, with the consequence of a greater distribution of values and a change in the annual average mean for the Greater London Authority (GLA). The variation is not a critical issue; as relative changes were being considered within each chapter. However, for future work the wider range of archetypes and their variants seen in chapter 6 onwards, being linked to the English Housing Survey (EHS) would be seen as more accurate and a better representation of the GLA domestic stock. The variations in exposure values for $PM_{2.5}$ for occupants seen are dependent on the scenario being investigated and therefore vary depending on occupant behaviour, activity and building parameters, as well as energy efficiency and ventilation measures.

By carrying out a scoping literature review of unintended consequences emanating from the application of energy efficiency measures on the housing stock and constructing a range of multizone ventilation and indoor environmental building stock models, combined with $PM_{2.5}$ modelling and energy analysis software; the research objectives (see Table 8.1 below) have been successfully achieved. The methodology has been shown to be effective for calculating both $PM_{2.5}$ (and other indoor domestic airborne pollutant) concentrations in different building archetypes subject to various energy efficiency and ventilation interventions, different locations, occupant groups, tenures and behaviours. This has shown the scope of impacts of this *unintended consequence* of policies to decarbonise the housing stock. The methodology developed here in conjunction with colleagues both at UCL and from other institutions can be used to investigate a variety of scales from individual properties to towns, cities or whole nation housing stocks where sufficient empirical inputs and stock data are available. This can assist policy makers, building designers and health professionals to recognise the impacts of energy efficiency and ventilation interventions and their impact on indoor concentrations of $PM_{2.5}$.

8.2 Research Limitations

This work uses modelling as the key means of future investigation in line with the quote in the acknowledgements at the start of this study: *"For some socio-technical systems, simulation is the only way we know of investigating their future states - If you do not trust a carefully executed simulation, you probably have less reason to trust anything else, including the way you currently make decisions."* Johnson, J (2001).

However, although empirical inputs to the modelling have been researched and used wherever possible, sensitivity analysis has been employed, and results have been compared to other modelling and empirical studies; they are, in the end, models, - a reflection of reality and not the reality itself. Modelling analyses such as this study rely on multiple assumptions and many uncertainties. The results should therefore be interpreted with a degree of caution as they are dependent on the range of assumptions and input parameters specified should therefore be seen only as indicative and relative rather than as precise calculations of impact. For example, and as previously stated, field studies show high variability in $PM_{2.5}$ emission rates from cooking: $2.4\underline{+}2.1 mg.min^{-1}$ (He *et al.*, 2004), $36\underline{+}98 mg.min^{-1}$ (Olson and Burke, 2006), $1.6\underline{+}0.6$ $mg.min^{-1}$ (Ozkaynak et al., 2006). Similarly, there are variations in deposition rate calculation methodologies with differing interpretations of surface area (Fogh *et al*., 1997; Thornburg, *et al*., 2001), which could lead to differences in absolute $PM_{2.5}$ exposures. The sensitivity analysis indicates that these large uncertainties in $PM_{2.5}$ emissions and deposition rates influence exposure. Although these would not generally affect the direction of change (lower $PM_{2.5}$ values in the 2050 scenario where PPV is used in conjunction with energy efficiency interventions), they may affect its magnitude, either positively or negatively. Further details of the uncertainty within the models and assumptions are shown in detail in Appendix E.

Comparisons of results for indoor annual $PM_{2.5}$ concentrations with both empirical studies (section 5.3.1) and other modelling studies (section 5.3.1) show that current day modelled exposure profiles are broadly consistent with published data, however there are some variations in $PM_{2.5}$ concentrations between the locations, possibly due to differences in construction geometries, building techniques and materials used. There is currently limited observed data on the impacts of retrofitting strategies on indoor air quality in general, with the health impacts of $PM_{2.5}$ in particular to compare against model outputs.

In both CONTAM and EnergyPlus as with all multizone modelling software, there is an inbuilt assumption that the air and therefore the contaminant is uniformly mixed within a zone. This prevents spatial variation occurring within zones, whereas in reality any individual close to a source (e.g. the cook) may be exposed to a higher concentration for a period such as a cooking event. Conversely, other members of the household may receive lower than estimated exposure. A simple CFD component (software add-on) has more recently been developed for CONTAM as well as a method for two dimensional modelling for zones such as hallways in order to address these issues (Wang et al., 2010). Due to the the complexities of applying these to a large number of buildings in the modelled building stock, along with the additional computer processing time currently required, CFD has not been used in this study. The consequence is that $PM_{2.5}$ exposures for occupants such as cooks, may have been underestimated.

There are substantial benefits to using a modelling in research. There is of course the obvious time-cost benefit compared to large-scale monitoring campaigns, but the main value of modelling is the ability to consider the impact of various changes/scenarios, whilst controlling for other variables – something that often cannot be done in reality. This includes the ability to explore possible impacts of future policies and can in theory help to produce guidance to assess/avoid some unintended consequences. Modelling could aid for example, the formulation of hypothesis and intervention strategies, which must of course be then tested in the field. However, although modelling is a cost effective option, it is also subject to large input uncertainties as related above. In this sense it is not a substitute for empirical data, upon which modelling is reliant anyway.

One aspect not factored into this study is the built-costs of the proposed interventions and their relationship to possible health gains/benefits. It may well be that the gains are not compatible with the outlay for all the different interventions described. Where economic cost is a key driver, a selection process of 'suitable' interventions or none, is likely to occur. Nonetheless, despite these uncertainties, the results provide important indications of likely impacts that can be used to help inform policy decisions and areas for further research.

8.3 Key Findings in Relation to Stated Objectives

The research outcomes of this thesis can be used to both inform and predict the consequences and causes of changes in indoor domestic $PM_{2.5}$ concentration arising from a variety of factors and can be used on a range of scales from single property to national housing stock. Key findings in relation to the stated objectives in section 1.3 are shown in table 8.1

Table 8.1 Key findings in relation to stated objectives

No.	Objective	Key Findings
1	Via a literature review, scope the causes and domains of impact of unintended consequences resulting from policies promoting the application of a variety of energy efficiency measures to housing.	The scoping review of impacts of policies to improve the energy efficiency of the UK housing stock, show a range of over 100 unintended consequences impacting a range of domains. Possible unintended consequences are related both to faulty policy formulation and to problems with implementation. In complex systems such as housing, policy formulation processes that focus on single or limited objectives, while taking inadequate account for the complex and dynamic inter-relationships between objectives and outcomes, are vulnerable to policy failure and negative unintended consequences such and changes in IAQ and specifically changes in indoor domestic PM2.5 concentrations. Multiple trade-offs (for example between emissions reduction and public health) may occur if the current policies are rigidly enforced as they stand. A research paper resulting from this aspect of the investigations was published in a peer reviewed journal and was awarded Indoor and Built Environment, Best Paper 2014 by Sage.
2	Investigate the range of methods used for modelling indoor $PM_{2.5}$ and the factors influencing exposure. Evaluate previous works and its applicability to the thesis questions.	Having investigated the different model types and chosen CONTAM/ EnergyPlus as the platforms for this study, comparison of modelled outputs with available empirical studies and previous published modelling work show comparable output values for both $PM_{2.5}$ and other pollutants. The models and the tools constructed from the models have been shown to be a useful for studying the relative impact of mitigation measures and their impacts on indoor pollutant concentrations and consequent health impacts on housing. The methodology is transferable to any location where sufficient empirical inputs and data sources are available and can be used at a range of scales.
3	Using CONTAM and modelling the current Greater London Authority (GLA) housing stock, investigate the application of a specific energy efficiency intervention (air-tightness with MVHR). Calculate post intervention $PM_{2.5}$ concentrations and quantify impacts for the GLA. Post process results for different occupant behaviours and activities.	Modelling shows the potential for considerable variation in GLA $PM_{2.5}$ exposure levels among household members, mainly due to proximity to source during emission periods. Present day high exposures for cooks from $PM_{2.5}$ emissions during cooking in domestic environments are avoidable through a comparatively simple adaptation such as the introduction and use of extraction equipment or by properly fitted, maintained and operated MVHR systems. These also help to remove some of the indoor $PM_{2.5}$ derived from environmental tobacco smoke, which would be beneficial for non-smokers in such dwellings.

No.	Objective	Key Findings
4	Investigate the key influencing variables on indoor $PM_{2.5}$ concentrations and uncertainty within the CONTAM models and applied at stock-level using differential sensitivity for both the input variables and computational processes.	Sensitivity analysis on the models input variables in chapter 5 highlights $PM_{2.5}$ emission sources (primarily cooking and smoking), ventilation behaviour (particularly window opening) and deposition as the largest variables influencing indoor concentrations within the models. However, it should be stressed that there is variation in the uncertainty of the individual parameters used. For example, variability in building height and volume is better understood than variability in behaviour and window opening and that there are large uncertainties in $PM_{2.5}$ emissions and deposition rates which influence exposure.
5	Using The SCRIBE tool (incorporating CONTAM) and modelling the current London and Milton Keynes housing stocks; apply a variety of energy efficiency and ventilation interventions and, investigate the potential for achieving climate change targets and the impacts on indoor $PM_{2.5}$ concentrations in two different locations.	Potential energy savings and health impacts are location specific and primarily driven by external conditions, the existing stock profile, building age and efficiency level - as seen in chapter 6 - with the greatest gains generally available with older building stocks, although occupant behaviour also plays a crucial role In the cities studied (i.e. London and Milton Keynes), the range of energy efficiency measures applied (including cavity or solid wall insulation, loft insulation to 250 mm, Installation of condensing boilers and central heating, draught stripping and fitting of new double glazing with trickle vents) were insufficient to reach the target of an 80% reduction in CO2 emissions from housing by 2050 relative to 1990 values. Changes in occupant behaviour or a combination of policies including decarbonisation of the electric grid maybe a way forward. Following interventions there are differences in CO2 emissions and in the internal concentrations of $PM_{2.5}$, -and other pollutants, such that blanket policies may not work. The exclusion of PPV from any retrofit scheme means that greater CO2 savings are made at the expenses of health as although outdoor $PM_{2.5}$ penetrating the building envelope is reduced, there are substantial increases in indoor source $PM_{2.5}$–and other pollutants- resulting in negative health impacts.
6	Using EnergyPlus and its GCM model building archetypes representative of the current and post retrofit English Housing stock to predict indoor $PM_{2.5}$ exposures from both indoor and outdoor sources. Using statistical analysis (ANOVA) to investigate differences between the various tenancies and income groups.	Modelling has shown that there are potential social inequalities arising as a result of the application of energy efficiency interventions and that these impact on current and future $PM_{2.5}$ indoor domestic exposures for different social economic groups. On average, all types of households below the low income threshold (LIT) experience greater overall concentrations of $PM_{2.5}$ than those above the LIT leading to consequences for occupant health. Below LIT income properties are generally shown to be more vulnerable to increased levels of outdoor $PM_{2.5}$ due to their location and indoor $PM_{2.5}$ from indoor sources, with $PM_{2.5}$ from cooking being the main source.

8.4 Novel Contributions

My study has used a series of simulations to investigate the changes in indoor domestic exposure to $PM_{2.5}$ resulting from of the application of energy efficiency and ventilations measures on the UK housing stock. These were subsequently used to ascertain the impacts other airborne pollutants in the UK as well as investigate locational impacts, occupant behaviour and effects on different income groups. It required a multi-disciplinary approach using, micro-environmental pollutant modelling and energy analysis software. This represents the first time a thorough investigation of the multiple influences on indoor domestic $PM_{2.5}$ concentrations have been carried out in the context of climate change and mitigation measures and fills an important research gap. The major contributions for which the author is responsible covers contributions to methods/tools and direct contributions to knowledge.

8.4.1 Contributions to Knowledge

- The first holistic review to characterise and enumerate the range and domains of impact of policies to reduce end-use housing energy demand in housing in the UK and the unintended consequences arising from such policies.
  As described in Chapter 2 and '100 unintended consequences of policies to improve the energy efficiency of the UK housing stock' published in Indoor and Built Environment. **Awarded Best paper 2014: Sage Publishing**

- The creation in conjunction with various team members of an enhanced, transferable methodology for modelling domestic stock profiles enabling the production of multiple geometries for building and systems able to comply with changing Building Regulations. These models enable investigation of a variety of future mitigation measures and the concentrations of $PM_{2.5}$ and other airborne pollutant types in multiples locations and at various scales. Additionally, an algorithm that post-processes $PM_{2.5}$ concentrations to yield individual occupant exposure based on activity and behaviour.

- The production of complex models in CONTAM representing the English housing stock. Other team members linked these to the English Housing Survey, such that they are able to predict occupant exposure to $PM_{2.5}$ and other airborne pollutant concentrations using a variety of energy efficiency and ventilation interventions and currently used within the *Department of Energy and Climate Change 'Health Impacts of Domestic Energy Efficiency' (HiDEEM) model (Hamilton et al., 2012, 2015), to monetise the health impacts of energy efficiency interventions.*

- The execution of differential sensitivity analysis on the housing stock modelling in order to quantify the key variables impacting indoor domestic $PM_{2.5}$ concentrations within the models and their distribution and thereby fill and important knowledge gap in modelling research.
  As described in Chapter 4 and 'Indoor $PM_{2.5}$ exposure in London's domestic stock: modelling current and future exposures following energy efficient refurbishment' published in *Atmospheric Environment.*

- A simple post-processing method for determining the relative exposures to $PM_{2.5}$ experienced by individual occupants' dependant on their location and activity within a property, highlighting the impact of differences in behaviour.
  As described in Chapter 5 and 'Indoor $PM_{2.5}$ exposure in London's domestic stock: modelling current and future exposures following energy efficient refurbishment' published in *Atmospheric Environment.*

- Additional enhancements to the 'SCRIBE' model produced by the author and other team members including the ability to simultaneously investigate various grid decarbonisation with energy efficiency measures on the domestic stock while quantifying the health impacts of $PM_{2.5}$ exposures and of that from other pollutants, end use energy demand and CO_2 emission reductions of a variety of possible policy options.

- Research filling a knowledge gap enabling quantification of possible changes in future population $PM_{2.5}$ exposure in indoor domestic environments, with a range of other airborne pollutants and subsequent health impacts, by modelling current and possible future stock profiles under a variety of mitigation scenarios for England.
  As described in Chapters 4, 5 and 'Multi-objective methods for determining optimal ventilation rates in dwellings' published in Building and Environment.

- The creation of $PM_{2.5}$ exposure profiles able to be used in Health Impact Assessment (HIA) software to quantify the health outcomes of the various strategies.
  As described in Chapters 5,6 and 7 and 'Health effects of energy efficiency interventions in England: a modelling study' published in BMJ Open.

- Quantification by modelling of the relative impacts of different urban locations and associated stock compositions, on indoor $PM_{2.5}$ and other pollutant concentrations in the housing stock, including the enhancement of the 'Strategies for Carbon Reduction In the Built Environment' (SCRIBE) tool to incorporate housing data for Milton Keynes enabling study at a local rather than purely regional scale. This study also combined energy efficiency and grid decarbonisation studies for the first time

As described in Chapter 6 and 'a tale of two cities: comparative impacts of CO2 reduction strategies on dwellings in London and Milton Keynes' published in Atmospheric Environment.

- Producing research linking income, tenancy and the impacts of energy efficiency interventions on indoor $PM_{2.5}$ exposures in homes via modelling, thereby pointing to a possible future area for empirical investigation.
  As described in Chapter 7 and 'Impacts of energy efficiency retrofitting measures on indoor $PM_{2.5}$ concentrations across different income groups in England: a modelling study. Published in Advances in Building Energy Research.

8.5 Recommendations

Based on the research conducted in this thesis, a number of recommendations can be put forward, some of which are for policy makers responsible for housing and energy efficiency/climate change objectives as well as some suggestions for future research.

1. In respect of policymakers, the study of unintended consequences and in particular the investigation of airtightness has demonstrated that the linkages identified in the literature form complex inter-relationships between various domains. This suggests that more holistic, multi-disciplinary approaches are needed to formulate and implement policies regarding housing rather than limited focus policies that may lead to unintended consequences. These include changes in IAQ and in reference to this study, changes in $PM_{2.5}$ concentrations and the health impacts that follow.

   It appears that the initial cause of such unintended consequences may lie in the indirect drivers such as policy conception in isolation from other policies (silo thinking) and the multiple goals for housing and their impacts. Problems occurring in delivery and application all flow from this point.

   It is suggested that – despite some efforts to include behavioural insights – the lack of integration with other goals around housing resulted in a misconception of the boundary of the problem. In addition, a mismatch exists between policy and the latest research and data, with no current mechanism to feed this back into policy-making (circular policy). It is suggested that linear, primarily single-focus policies in a complex dynamic environment are by nature problematic, as the dynamic complexity requires a different approach. It is recommended that as a starting point, one possible way forward for future policy construction involves the use of a widely-applicable collaborative mapping and simulation method (participatory system dynamics modelling) with representatives from organisations with a stake in housing policies (sectors of national and local government, non-government organisations, construction and housing industries, user's representatives and cross-disciplinary researchers). This method can be scaled to facilitate interdisciplinary knowledge generation and/or to include scenario analyses and modelling.

2. For policymakers, modelling has shown that there are potential social inequalities arising as a result of the application of energy efficiency interventions and that these impact on current and future $PM_{2.5}$ indoor concentrations. Following empirical monitoring and if the model findings are shown to be correct; it is suggested that homes with incomes below the low income threshold (LIT) should be prioritised for refurbishment for better IAQ and energy efficiency-especially those that are not owner-occupied. Schemes should also take into account their location which are often in areas of high external $PM_{2.5}$ levels when determining ventilation strategies.

3. For designers and those involved in the formulation and application of the Building regulations, this study shows that in order to ensure maximum health co-benefits from energy efficient refurbishment, purpose provided ventilation (PPV) needs to be included in any scheme. Although the current regulations for new build properties are clearly stipulated in Approved Document F (ADF) of the Building regulations, it is questionable whether refurbishments always comply with this. For example, the final decision regarding the choice of the inclusion of trickle vents in double glazing is left to the householder. In addition, where interventions are carried out independently over a period of time, the realisation of increased airtightness may not be apparent. This study has shown that where airtightness reduces end-use energy demand and PPV is included, the additional energy needed for space-heating is minimal but the health gains due to a reduction in indoor $PM_{2.5}$ concentrations are potentially large. If high external levels of $PM_{2.5}$ exist, then any PPV will require filtration. Changes to ADF or rather checks on compliance may need to be a statutory requirement to ensure building retrofits comply.

4. In order to obtain both health gains (e.g. due to reductions in $PM_{2.5}$ concentrations) and promote success in achieving CO_2 emission reduction targets in different locations, policymakers may need to consider a wider view that includes strategies to extensively decarbonize the electricity grid and possibly other policies with a move away from the reliance on residential use of gas combined with the application of energy efficiency measures on the built stock.

5. $PM_{2.5}$ is one of many airborne pollutants. When designing strategies for wellbeing in buildings, optimal solutions are needed that take into account the range of outdoor and indoor pollutants sources that occupants are exposed to as well as temperature changes.

8.6 Suggested Future Work

This thesis has provided a springboard for further research into the changes in exposure to $PM_{2.5}$ in dwellings resulting from energy efficient refurbishment of the housing stock. While this initial examination has raised a number of issues, there are suggested future areas of investigation.

1. The need to adapt to and mitigate climate change impacts requires a sense of urgency in developing policies influencing England's housing, which currently appear to be lacking. As a starting point, one possible way forward for future policy construction involves the use of a widely-applicable collaborative mapping and simulation method (participatory system dynamics modelling) with representatives from organisations with a stake in housing policies (sectors of national and local government, non-government organisations, construction and housing industries, user's representatives and cross-disciplinary researchers). Such studies would need to specifically address the issue of IAQ and indoor $PM_{2.5}$ exposures along with other priorities.
2. There is a need for large scale pre and post-retrofit investigations of properties to measure the impact on $PM_{2.5}$ concentrations (and other pollutants) of the application of energy efficiency to confirm, adapt or refute the findings of this study. In addition to physical measurement it is important that interviews and questionnaires (pre and post occupancy evaluations) are carried out to integrate occupant behaviour with the physical science to fully understand all influences on indoor $PM_{2.5}$ exposures.
3. As yet, this study did not consider the cost element of suggested interventions, or a cost comparison for grid decarbonization scenarios. Combined research with an environmental economist would be a useful addition to this study and quantify the most cost effective options and the range of costs for various interventions. Being able to consider the costs involved with the application of the proposed energy efficiency and ventilation measures could be contrasted with the cost savings from long term health benefits would be useful for policy makers. Whilst the HiDEEM model does consider this, access to actual construction costings would prove useful.
4. Although this study has compared different locations (Milton Keynes/London), with minimal additional development of the HiDEEM/SCRIBE model, a useful investigation comparing the different government regions in England could be carried out, which would help highlight priorities for refurbishment to both maximise health gains from increased indoor temperatures reduced indoor pollution exposure and would help promote energy savings.
5. Additional pollutants (such as Volatile Organic Compounds (VOCs), Carbon Monoxide(CO) and Nitrogen Dioxide (NO_2) need to be investigated and added to the CONTAM and other models in order to ensure that all indoor airborne pollutants are covered in the modelling. Taken together with existing temperature changes, this will enable optimal strategies for energy efficiency and ventilation strategies to be suggested and investigated.

6. Due to the wide variety of non-domestic building architypes, material use and construction techniques, few studies have investigated such properties with the same depth of analysis seen here. Characterisation and modelling analysis of these would be a logical next step for this study.
7. This study separated the outdoor and indoor sourced components of $PM_{2.5}$ due to the ongoing debate over their relative toxicity. This is an area that could be studied in collaboration with toxicologists. Also, potentially adjusting exposure-response coefficients to account for any differences in indoor/outdoor exposure in collaboration with epidemiologists. A recently submitted research paper to which the author contributed (Milner et al., submitted 29th June 2017), investigates the exposure mortality relationship for $PM_{2.5}$ from outside sources in domestic properties this represents an initial in this investigation.
8. Modelling analyses such as this study rely on multiple assumptions and model verification is needed for the uncertainties. Its results should therefore be interpreted only as indicative and relative rather than as precise calculations of impact. Nonetheless, despite these uncertainties, the results provide important indications of likely impacts that can be used to inform policy decisions. However, there is currently limited observed data on the impacts of retrofitting strategies on indoor air quality in general and in $PM_{2.5}$ concentrations and health in particular in order to compare against model outputs. There is a need for large scale pre and post retrofit empirical monitoring studies to provide model verification as well as more accurate modelling inputs in some instances, particularly those relating to occupant behaviour. It is suggested that this should be an urgent priority for further investigation. Further uncertainty/sensitivity analysis is required enabling a disentangling of the relative importance of inputs required for the building-level model, and inputs required for the scaling up of the building-level models. A possible route forward would be via ensemble analysis.
9. As previously stated, the modelling tools used assume a homogenous mixing of the air and therefore the airborne pollutant, which may incorrectly estimate exposures, especially for those (such as cooks), who are closest to the immediate source of $PM_{2.5.}$ As computational power and speed increases, it is suggested that the use of CFD techniques in combination with the data used in this study, could aid a more accurate assessment of individual $PM_{2.5.}$ exposure and consequent health impacts for individual occupants.

In addition, the models used in this study were constructed such that a further range of pollutants could be investigated in various scenarios. Further studies not directly related to this thesis, but using these models or variants of them have been carried out and are detailed in Appendix G.

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Appendices

Supplementary Material

Appendix A

Range of Key Words for Investigation of Unintended Consequences

Key words for investigation

Adaptation
Adaptive behaviour
Air permeability
Air pollutants
Air pollutants/adverse effects
Air pollution/pollutants
Air pollution/pollutants indoors
Air quality
Air quality/health impacts
Air conditioning
Attention
Asthma
Balanced ventilation
Behavior/behaviour
Behavior/behaviour change
Blood pressure
Body temperature
Building
Building codes
Building fabric
Building materials
Building performance
Building physics
Building regulations
Built environment
Carbon dioxide
Carbon emissions
Carbon intensity
Carbon reduction
Cardiovascular Diseases/s
Central heating
Climate change/ impacts
CO2/reduction/emissions
Cohort studies
Cold/homes
Comfort
Community
Demand response
Diet/adverse effects
Domestic/property
Disease/causes/ transmission
Draft proofing
Drying/out
Dust mites
Dwellings
Energy Company Obligation
Emission/s/rates
Energy/assessment
Energy company obligation
Energy conservation
Energy consumption/demand
Energy efficiency/measures
Energy efficiency strategy
Energy efficient design
Energy performance/policy
Energy saving/use
Environment/design
Environmental exposure
Environmental impact
Environmental pollution
Environmental monitoring
Exposure
Exterior insulation
External wall insulation
Fuel consumption
Fuel poverty
Glazing /double
Green building/s
Green deal
Green growth
Greenhouse effect/gases
Greenhouse warming
HDM/house dust mite/s
Health/behavior/behaviour
IAQ
Indoor air quality
Indoor environment/health
Indoor temperature
Infection/transmission
Inhabitants
Inhalation/exposure
insulation
Jevons
Lifestyle
Low carbon/materials
Low carbon energy/homes
Low carbon initiative
Low carbon refurbishment/s
Low temperature/s
Mechanical ventilation
Mental health/impacts
Moisture/damage/transfer
Mortality
Mould/growth
MVHR/ heat recovery
Natural ventilation
Noise/health impacts/effects
Obesity/metabolism
Occupant/s behaviour
Occupant /s/control/ health
Occupant/s response
Outdoor air/pollution
Overcrowding
Overheating
Particulate matter/PM2.5
Passivhaus
Perception/s
Performance
Planning
Policy/making/measures
Pollution

Complex dynamic systems	Health impacts	Post occupancy/ evaluation
Compliance	Heat/emission/s	Poverty
Condensation	Heat recovery	Power/plants
Consumer behavior/ behaviour	Heritage	Psychological health
CONTAM	HMO	Psychological wellbeing
Controls/usability	Houses of multiple occupation	Public health
Crime	Homes/housing/household	Radon
Decarbonisation	Housing/crowding/improvements	Rebound/effect
Decentralised/energy	Housing retrofit	Refit/refurbishment
Demand control	humidity	Regulation/s
Relative humidity/RH	Stress/psychological	Thermal performance
Renewable/energy/heat	Sudden infant death/SID	Traditional buildings/dwellings
Research gaps/needs	Summer temperatures	Ultra violet rays/ UVA/UVB
Residential/effects/energy use	Sunlight	Uncertainty
Residential ventilation	Supply chain	Unintended/ consequences
Respiratory health	Sustainable/ sustainability	User behavior/behaviour
Retrofit/ retrofitting	Sustainable construction	User controls/s
Risk assessment	Sustainable consumption	UV light/rays
Satisfaction	Sustainable homes	Ventilation
Sick building syndrome/SBS	Systematic review/s	Vitamin D/deficiency/effects
Sick leave	System dynamics	Volatile organic compounds/VOC
Sleep/disorders/causes	Systems approach	Wall/s
Social conditions/housing	Take-back/temperature	Water penetration
Socio-economic/outcomes	Temperature	Welfare/wellbeing
Socio-technical	Tenure	Window/opening/behaviour
Space heating	Thermal comfort	Winter
Strategy/strategies	Thermal insulation	

Appendix B

Additional Unintended Consequences

and References

*For No 1-12 see section 2.3.1-Table 2.3

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
12	Airtightness with PPV (no heat exchange)	Lower or equivalent air change rate	Changes in indoor air quality (IQA)	Reduction in most indoor sourced pollutants/increase in externally sourced if not filtered.	Physical Health	+/-	Milner et al., 2012. Shrubsole et al., 2012. **B,C,D,E,F**
13	Airtightness with PPV (no heat exchange)	Increased air change rate.	Changes in indoor air quality (IQA)	Reduction in most indoor sourced pollutants/ Increase in externally sourced if not filtered. Variation with pollutant type	Physical Health.	+/-	Milner et al., 2012. **B,C,D,E,F**
14	Airtightness with PPV (no heat exchange)	Increase in air change rate	Over ventilation	Energy efficiency gains underwritten by ventilation heat loss. GHG emissions unaffected or increased. Increased fuel bills	Environment. Household finances	–	Milner et al., 2012. Review of ADF2010 and BS 5250 **B,C,D,E,F**
15	Airtightness with PPV (no heat exchange)	Balanced air exchange	Changes in indoor air quality (IQA)	Reduction in indoor sourced pollutants and externally sourced if filtered. Health impacts are generally positive	Physical Health. Environment.	+	Wilkinson et al., 2009. Shrubsole et al., 2012. Milner et al., 2012. **B,C,D,E,F**
16	Airtightness with MVHR and filtration	Balanced air exchange	Changes in indoor air quality (IQA)	Reduction in indoor sourced pollutants	Physical Health. Environment	+	Wilkinson et al., 2009. Milner et al., 2012. **B,C,D,E,F**
17	Airtightness with MVHR and filtration	Poor specification or Installation	Noise, disturbed sleep, annoyance.	System switched off, poor IAQ, reduction in health gains, reduction in energy efficiency gains. Decreases in GHG emissions	Physical health Psychological Well Being Environment.	+/-	Balvers et al., 2012 **B,C,D,E,F**
18	Airtightness with MVHR and filtration	Poor upkeep blocked filters, failure.	Low air change Quieter Environment	Increase in indoor pollution levels. Microbiological growth. Increased fuel bills Increased GHG emissions	Physical health. Household finances Environment	+/-	Thorpe 2011. Balvers et al., 2012. **B,C,D,E,F**
19	Airtightness with MVHR and filtration	Quieter Environment		Reduction in household accidents. Grades in school show an improvement. Mental Alertness/ increase in concentration	Physical health. Mental health Psychological Well Being	+	Mendell & Heath 2005; Mumovic et al., 2009. **D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
20	Introduction of Efficient Technologies	Poor grasp of new technology/ equipment	Increased energy use or inefficient use.	Energy efficiency gains lost. Possible increase in GHG emissions. Reduction or increase in health benefits.	Environment Physical health	+/-	Rossau et al., 2011. Balvers et al., 2012; Mulligan and Broadway, 2012 **B,C,D,E,F**
21	Introduction of Efficient Technologies	Adaptive approach to system design	Increased occupant interaction with system	Adaptation of environmental conditions, increase in comfort	Physical and Psychological wellbeing Behaviour	+/-	Wagner et al., 2007 ; Gupta et al.,2010; Toftum, 2010 **B,C,D,E,F**
22	Introduction of Efficient Technologies	Centrally or controlled system approach	Little possible occupant interaction with system	Decreased comfort levels, greater instances of SBS	Physical and Psychological wellbeing	–	Toftum, 2010; Fabi et al., 2012 **B,C,D,E,F**
23	Introduction of Efficient Technologies	Centrally or controlled system approach	Little possible occupant interaction with system	Reversion to window opening to control environment	Physical wellbeing Environment	–	Sharpe and Shearer, 2012 **B,C,D,E,F**
24	Introduction of Efficient Technologies	Occupant interaction with systems / elements	Trickle vents closed to prevent drafts/ heat loss	Decrease in air change rate, increase in indoor pollution levels. Decrease in energy use/ GHG emissions	Physical health Environment	+/-	BRE, 2005 **D,E,F**
25	Introduction of Efficient Technologies	Imposing smart meter systems without consultation	Reaction against “surveillance” refusal to cooperate.	Lack of accurate monitoring data on energy use in homes, no measure of energy saving policy success.	Environment	–	Infowars, 2012 **B,C,D,E,F**
26	Introduction of Efficient Technologies	Inappropriate Use/application to structure	System failures	Disillusionment with technology Bad publicity, impact on uptake	Environment Economics	–	Stein et al.,2010 **B,C,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature: Unintended Consequence	Domain	+/-	Reference
27	Insulation	Warmer Environment	Greater room availability. Changes in room occupation patterns	Fragmentation/Cohesion Family dynamics. Changes in home/work relationships.	Physical and wellbeing and behaviour	+/-	Sanz et al., 1993; Van Kempen et al., 2012. Shrubsole et.al 2012 **B,C,D,E,F**
28	Insulation		Warmer Environment	Thermal imaging ineffective. Cultivation of Drugs (Marijuana) go undetected by this means	Legal Physical health	–	Johnson and Miler, 2011; Gibson, 2012 **E,F**
29	Insulation		Warmer Environment	Take back for comfort: "Jevons" effect. Increased fuel use and GHG emissions despite improvements	Physical Health Environment	+/-	Davies and Oreszczyn, 2012. Deurinck 2012 **B,C,D,E,F**
30	Insulation		Warmer Environment	Increased time spent indoors Sedentary behaviour Weight gain /obesity	Physical Health	–	Johnson et al., 2011 **E,F**
31	Insulation		Warmer Environment	Increased time spent indoors. Reduction in social cohesion	Mental Health Psychological Well Being	–	Sanz et al., 1993. Van Kempen et al., 2012. **B,C,D,E,F**
32	Insulation		Warmer Environment	Reduction in winter mortality.	Physical Health	+	Mavrogianni, 2012 Oikonomou, 2012 **B,C,D,E,F**
33	Insulation	Poor design, increase in thermal mass	Higher average indoor temperatures	Summer overheating. Considered uninhabitable Breach of duty, Defective Premises Act 1972. MM vs Western Homes 2011	Physical Health Legal	–	Mavrogianni,2012 Oikonomou, 2012 **D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
34	Insulation	Poor design, increase in thermal mass	Higher average indoor temperatures	Additional cooling equipment used in summer increased energy use/ GHG emissions	Physical Health Environment	–	Energy Savings Trust, 2008a, b. Beizaee, 2013 **B,C,D,E,F**
35	Insulation		Warmer Environment	Increase in severity of skin infections, bed bugs, reactions to allergens	Physical Health	–	Ucci et al., 2011 **E,F**
36	Insulation		Warmer Environment	Attraction of pests and vermin, spread of disease, destruction of property	Physical Health	–	Shelter, 2012 **D,E,F**
37	Insulation		Higher average Indoor temperatures	Increase of VOCs from Building materials and fittings. IAQ changes	Physical Health	–	Zhang et al., 2007 Xiong and Zhang 2010 **D,E,F**
38	Insulation		Higher average Indoor temperatures	Increased frequency of eating breakfast and dinner at home. Decreased prevalence of obesity	Physical Health	+	Mavrogianni et al., 2003 **D,E,F**
39	Insulation		Higher average Indoor temperatures	Reduced energy required to maintain body temperature. Increased prevalence of obesity	Physical Health	–	Mavrogianni et al., 2003 **D,E,F**
40	Insulation		Higher average Indoor temperatures	Reduced cold-induced "comfort-food" intake. Reduced prevalence of obesity	Physical Health	+	Mavrogianni et al., 2003 **D,E,F**
41	Insulation		Higher average Indoor temperatures	Cost per head of home heating reduced. Increased financial control Reduced stress induced comfort eating. Reduced prevalence of obesity	Physical Health Family Economic Psychological wellbeing	+	Marmot, 2011 **B,D,E,F**
42	Insulation		Higher average Indoor temperatures	Decrease in fuel bills. Increased micro nutrient levels based or released funding spent on quality food	Physical Health Psychological wellbeing	+	Mavrogianni et al., 2003 **D,E,F**

No	Policy Impact on Buildings		Impacts on People/ Nature		+/-	Reference
			Unintended Consequence	**Domain**		
43	Insulation	Higher average Indoor temperatures	Greater mobility/ dexterity for arthritis sufferers	Physical Health	+	Boardman et al, 2010 **B, D,E,F**
44	Insulation	Higher average Indoor temperatures	Increase in bedroom temperature above 21°C linked to improved mental health across life time	Mental Health	+	Marmot, 2011 **B,D,E,F**
45	Insulation	Higher average Indoor temperatures	Improvement in adolescent mental heath	Mental Health	+	Marmot, 2011 **B,D,E,F**
46	Insulation	Higher average Indoor temperatures	Increase in immunity and decreases in multiplication of common cold	Physical Health	+	Marmot, 2011 **D,E,F**
47	Insulation	Higher average Indoor temperatures	Less time off due to cold related sickness Higher productivity	Physical Health Economic	+	Marmot, 2011 **D,E,F**
48	Insulation	Higher average Indoor temperatures	Reduction in injuries in the elderly or infirm = reduction in NHS costs due to reduced number of admissions	Physical Health Economic	+	Boardman, 2010 **B,D,E,F**
49	Insulation	Higher average Indoor temperatures	Older people less likely to require residential care. Decrease of financial burden	Physical Health Economic	+	Boardman, 2010 **B,D,E,F**
50	Insulation	Higher average Indoor temperatures	Reduction in need for winter fuel payments.	Economic	+	Tod et al., 2012 **B,D,E,F**
51	Insulation	Higher average Indoor temperatures	Infant weight gain improved and development status	Physical Health	+	Liddell, 2010 **B,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
52	Insulation		Higher average Indoor temperatures	Increase fire risk from old electrical wiring and foil backed insulation	Physical Health Environment	-	HIP Report, 2014 **E,F**
53	Interior insulation to walls	Outer skin subject to wider temperature fluctuations	Increase freeze thaw, frost shattering on brickwork. Loss of envelope integrity	Damage and loss of appearance of cultural heritage. Resource use to repair damage.	Environment Psychological Wellbeing	-	BS EN771-1 BRE Digest 369 BS 5250 : **B,C,D,E,F**
54	Interior insulation to walls	Creation of inner thermal envelope	Interstitial/inner face condensation, unseen until severe	Increase in RH, Mould-microbiological growth Increase in HDM, severity of asthma and allergies. Resource use to repair damage	Physical Health Environment	-	Ucci et al., 2011. Viitanen et al., 2010 **B,C,D,E,F**
55	Interior insulation to walls on traditional buildings with in-built joists	Thermal bridging	Focus and condensation of moisture on joist ends, Rotting of structural elements	Danger of structural collapse, Mould-microbiological growth Use of resources to rebuild /refurbish structures	Physical health Environment	–	STBA, 2012 **B,C,D,E,F**
56	Introduction of new components into traditional structures	EWI on traditional structures Differential movement	Envelope breach Moisture penetration, damage to building fabric	Increase in RH, Mould-microbiological growth Resource use to repair damage	Physical health Environment	–	Kunzel and Zirklebach, 2008 **B,C,D,E,F**
57	Introduction of new components into traditional structures	Inappropriate survey practices for EWI on traditional structures	Design flaws Thermal bridging	Condensation Moisture ingress and mould Resource use to repair damage	Physical health Environment	–	Hopper et al., 2012 **B,C,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
58	Introduction of new components into traditional structures	Roof space thermally sealed Colder roofs. Increased snow/ice loading	Possibility of structural damage to roof components/ guttering. Ingress of water into damaged structures	Moisture ingress and mould Resource use to repair damage	Physical health Environment	–	Rye, 2012 **B,C,D,E,F**
59	Introduction of new components into traditional structures	Underfloor space thermally sealed from dwelling area. High RH in crawl space	Condensation and undetected rotting of untreated joists in floor space.	Moisture ingress and mould. Structural failure Resource use to repair damage	Physical health Environment	–	Lstiburek, 2008 BS 8103-3:2009 **B,C,D,E,F**
60	Introduction of new components into traditional structures	Lack of clarity for historic buildings. Energy efficiency as main driver of change	Lack of consistency in planning policies. Application of inappropriate energy efficiency measures	Failure to achieve full energy improvements for this category of building. Increase in GHG emissions. Damage to heritage assets, disconnection from sense of history.	Environment Psychological Well Being	–	Powter and Ross, 2005; Friedman and Cooke, 2012. **B,C,D,E,F**
61	Introduction of new components into traditional structures	Use of imported fluorescent light bulbs	Exposure to UVA and UB via low ferrous glass	Health implications if light to close to skin or prolonged exposure, burns etc.	Physical health	–	Cantwell et al., 2009; Sharma et al., 2009 **C,D,E,F**

No	Policy Impact on Buildings		Impacts on People/ Nature		+/ -	Reference
			Unintended Consequence	**Domain**		
62	Double glazing fitted	Reduction in penetration of UVB UV, UVA wavelengths	Reduction in Vitamin D Production. Increases in low level diabetes, particularly in infants	Physical Health	–	Pittas et al., 2007 Zipitis and Akobeng, 2008 Ahmed et al., 2012 **B,D,E,F**
63	Double glazing fitted	Reduction in penetration of UVB UV, UVA wavelengths	Reduction in Vitamin D Production. Increases in cardiovascular disease.	Physical Health	–	Parker et al., 2010 Wang et al., 2010 Annuzzi et al., 2012 **D,E,F**
64	Double glazing fitted	Reduction in penetration of UVB UV, UVA wavelengths	Reduction in Vitamin D Production. Increase in type 2 diabetes and metabolic syndrome in middle aged to elderly populations	Physical Health	–	Parker et al., 2010 **D, F**
65	Double glazing fitted	Reduction in penetration of UVB UV, UVA wavelengths	Reduction in Vitamin D Production. Decreased physical performance. Increased risk of fracture and fails in elderly; particularly post-menopausal women with osteoporosis	Physical Health	–	Gaugris et al., 2005 **D,E,F**
66	Double glazing fitted	Reduction in penetration of UVB UV, UVA wavelengths	Reduction in Vitamin D Production. Increases in blood pressure and associated risks.	Physical Health	–	Pilz et al., 2009 **D,E,F**
67	Double glazing fitted	Reduction in penetration of UVB UV, UVA wavelengths	Reduction in Vitamin D Production. Increases in low level diabetes, particularly in infants	Physical Health	–	Pittas et al., 2007 Zipitis and Akobeng, 2008 Ahmed et al., 2012 **B,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
68	Double glazing fitted	Reduction in penetration of UVB UV, UVA wavelengths		Reduction in Vitamin D Production. Increases in cardiovascular disease.	Physical Health	–	Parker et al., 2010 Wang et al., 2010 Annuzzi et al., 2012 **D,E,F**
69	Double glazing fitted	Reduction in penetration of UVB UV, UVA wavelengths		Reduction in Vitamin D Production. Increase in type 2 diabetes and metabolic syndrome in middle aged to elderly populations	Physical Health	–	Parker et al., 2010 **D, F**
70	Energy Supply moved to in urban areas	Cheaper more efficient supply.	Possible Increase in local pollution load from biomass plants and associated transport.	Reduction in relative CO_2 emissions. Increase in health problems. Possible rebound effects Increase UHI effect.	Environment Physical health		Thorley, 2008, Quiggin, 2012 Davies and Oreszczyn, 2012. **B,C,D,E,F**
71	Energy Supply moved to in urban areas	Cheaper more efficient supply.	Reduction in relative CO_2 emissions	Local energy security, capacity building, jobs, financial improvements, hope	Psychological Well Being Environment Economy	+	Del Rio, 2008 **C,D,E,F**
72	Changes to energy supply and type	Micro turbines placed on buildings	Noise problems especially during turbulence.	Sleep issues. Local supply, reduction in relative CO_2 emissions	Physical health Environment	+/-	Baker et al., 2012; Rogers and Omar 2012 **B, C, D,E,F**
73	Changes to energy supply and type	Electricity becomes the dominant domestic fuel	Changes in indoor air quality (IQA)	Reductions in personal exposure to CO, NO_2 from gas. Reduction in RH (air feels drier). Fuel security relative to gas/oil supply (peak oil issues).	Physical health Economy Environment	+	Olson and Burke 2006; Dennekamp et al., 2010 **B,C,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
74	Changes to energy supply and type	Use of gas increases (fracking)	Changes in indoor air quality (IQA) Increase in RH	Increases in personal exposure to CO, NO_2 from gas. Increase in RH, mould, HDM.	Physical health		Milner, 2011; Ucci, 2011; Thomson, 2013 **D,E,F**
75	Take up of the Green Deal	Upgrade of dwellings	Increase in installation/ maintenance costs	Reduction in disposable income, stress, time pressures. Social determination of comfort. Heat or eat!	Physical health Psychological wellbeing Family economics	+/-	STBA, 2012 Thomson, 2013 **B,C,D,E,F**
76	Take up of the Green Deal	Upgrade of dwellings	Increase in installation/ maintenance costs	Social determination of comfort. Heat or eat!	Physical health Psychological wellbeing Family economics	–	Thompson, 2013 **B,C,D,E,F**
77	Take up of the Green Deal	Upgrade of dwellings	Increase in installation/ maintenance costs	Increase in rents Overcrowding HMO/ poverty. Increased exposure to pathogens. Infectious diseases.	Physical health Psychological wellbeing Family economics	–	Beggs et al., 2003; Noakes et al., 2007 **B,C,D,E,F**
78	Take-up of the Green deal and associated policies	Upgrade of dwellings		Increase in rent Overcrowding HMO/ poverty Long term effects on future socio-economic status and wellbeing.	Social cohesion Social mobility	–	Thorley, 2008, Quiggin, 2012 Davies and Oreszczyn, 2012. **B,D,E,F**
79	Take-up of the Green deal and associated policies	Upgrade of dwellings		Increase in rent Overcrowding HMO/ poverty. Negative Impacts on child development	Physical Health	–	Shelter, 2005 **E,F**
80	Take-up of the Green deal and associated policies	Upgrade of dwellings		Increase in rent Overcrowding HMO/ poverty Increase in Sudden Infant death Syndrome (SIDS)	Physical Health	–	Baker et al., 2012; Rogers and Omar 2012 **B, D, E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	**Domain**		
81	Take-up of the Green deal and associated policies	Upgrade of dwellings		Increase in rent, inability to pay, risk of homelessness	Physical health. Social cohesion	–	Olson and Burke 2006; Dennekamp et al., 2010 **B,D,E,F**
Policy framing and implementation issues and the unintended consequences on the range of framework domains							
82	Take up of the Green Deal	Limited cash back scheme. No product subsidy	Delay in go ahead for existing schemes or new schemes	Delay in achieving energy goals. Drop off in number of applications once scheme ends.	Economic	–	Federation of Master Builders, 2012 **B,C,D,E,F**
83	Take up of the Green Deal	Lack of effective marketing of the policy	Households failing to register for the deal.	Policy failure, low curbing of GHG emissions from domestic sector	Environmental	–	Independent, 2013 **D,E,F**
84	Take up of the Green Deal	Lack of clear legislation or publicity of incentives.	Mismatch between policies and goals	Reliance on voluntary public engagement “altruism”. Richer more likely to engage. Increase of gap between the rich and poor. Most needy don’t benefit from policy	Physical health Social Inequalities	–	Boardman, 2010; Crosbie, 2010 **E,F**
85	Take up of the Green Deal	Public resistance, hassle factor, economic uncertainty, finance	Households failing to embrace the deal	Policy failure, low curbing of GHG emissions from domestic sector	Physical health Environmental	–	Dawson, 2012 **B,C,D,E,F**
86	Take up of the Green Deal	Limited scope of available finance	Necessary Façade and building fabric repairs excluded from scheme	Damage to fabric and contents if schemes implemented. Failure to achieve Energy savings, moisture ingress, health	Building Physical health Environmental	–	Davies and Oreszczyn, 2012; STBA, 2012 **B,C,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
87	Take up of the Green Deal	Limited scope of available finance	Necessary Façade and building fabric repairs excluded from scheme	Additional costs cause delay or decision not to proceed with schemes. No reduction in GHG emissions, health.	Building Physical health Environmental	–	STBA, 2012 **B,C,D,E,F**
88	Take up of the Green Deal	Scope of available finance extended	Façade and building fabric Repairs included in financing	Achievement of goals beyond GD expectation. Preservation of structures and construction skills base. Improved forum for passing on experience and good practice	Building Physical health Environmental	+	Davies and Oreszczyn, 2012; STBA, 2012 **B,C,D,E,F**
89	Current Guidance and Regulation	Disconnect between best research and current guidance	Inappropriate material application on retrofit properties	Damage to fabric and contents. Failure to achieve Energy savings, moisture ingress, health	Building Physical health Environmental	–	Hens, 2002; STBA, 2012 **B,C,D,E,F**
90	Current Guidance and Regulation	Double glazed units specified against secondary glazing in historic buildings	Unnecessary increased heat loss. Conflict with conservation and aesthetic needs	Increased energy use. Increased GHG emissions. Non-optimum solutions specified	Building Environmental	–	Wood, 2009; Baker,2010; EST, 2011 **B,C,D,E,F**
91	Current Guidance and Regulation	Under estimation of U-values on solid walls	Insulation over engineered. Non-optimal solutions used	Possible reduction in winter morbidity /mortality. Summer overheating very likely. Uncertain energy outcomes and fabric impacts	Building Physical health Environmental	+/-	BR443 ; EN ISO 6946: 1997; Mavrogianni, 2012; Oikonomou, 2012; STBA, 2012 **B,C,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	**Domain**		
92	Current Guidance and Regulation	Lack of research on thermal bridging/ mass in traditional structures	Inappropriate material application on retrofit properties	Focus and condensation of moisture. Mould Rotting of structural elements. Cost of replacement of defective members	Building Physical health Environmental	–	Hens, 2002; STBA, 2012 **B,C,D,E,F**
93	Current Guidance and Regulation	No data base of in-situ U values for traditional walls	Inappropriate use of current regulations.	Appropriate interventions. Uncertainty over possible energy savings. Fabric protection. Alteration to BR443 and Rd299v 9.91 (Appendix S, 2012) to provide better modelling conventions.	Building Environmental	–	Rye, 2010, SPAB, 2012 **B,C,D,E,F**
94	Current Guidance and Regulation	Non systematic approach to heat loss in traditional buildings	Failure to take into account interactions between heat moisture and air.	Inappropriate refurbishment strategies for traditional building. Failure to achieve energy reduction. Moisture transfer issues	Building Environmental	–	STBA, 2012 **B,C,D,E,F**
95	Current Guidance and Regulation	Almost no information on heat loss via pre 1919 floors	Poor understanding of insulation impacts on floors in traditional structures	In appropriate application of materials, Condensation risk, mould. Slip/ fall hazards	Building Environmental Physical health	–	STBA, 2012 **B,C,D,E,F**
96	Current Guidance and Regulation	Use of BS 5250: 2011 for moisture risk "Glaser method"	No allowance made for hydroscopic sorption, or liquid transport, rain	Unclear guidance and possible inappropriate construction design and practice. Moisture, Energy and health issues. Suggested use of BSEN 15026:2007	Building Environmental Physical health	–	BS 5250: 2011, BS EN ISO 13788: 2002; Olof Mundt-Petersen, 2012 **B,C,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
97	Current Guidance and Regulation	Lack of data on ventilation and heat loss from traditional buildings	Possible misapplication of ventilation systems and strategies. MVHR	Health impacts from airborne pollutants	Physical health Building	–	STBA, 2012 Wilkinson et al., 2009. Das et al, 2013 **B,C,D,E,F**
98	Current Guidance and Regulation	Lack of data on ventilation and heat loss from traditional buildings	Possible misapplication of ventilation systems and strategies. MVHR	New generation of sick building syndrome (SBS) Need for Promotion of health and well-being on equal footing with CO2 reduction	Physical health Building	–	LCC: ICT; 2010 STBA, 2012 **B,C,D,E,F**
99	Current Guidance and Regulation	Technical or Product based approach used.	Issue of ventilation, IAQ and behaviour not linked in Green deal	Systematic approach needed to avoid multiple –ve health impacts e.g. from airborne pollutants and impacts on other building elements	Physical health Building	–	Wilkinson et al., 2009 ; Shrubsole et al., 2012; STBA, 2012 **B,C,D,E,F**
100	Current Guidance and Regulation	Element based solution used	Improvement of single element only e.g. window	Issues created e.g. thermal bridging, changes in ventilation rates, IAQ issues	Physical health Building	–	STBA, 2012 Das et al., 2013 **B,C,D,E,F**
101	Current Guidance and Regulation	Technical or Product based approach used	Unrealistic expectations e.g. low U-value targets for wall	Thermal bridging Commercial guidance for product values. Whole house heat loss incorrectly calculated	Building Environmental	–	STBA, 2012 **B,C,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
102	Current Guidance and Regulation	CO_2 reduction and economic considerations as main drivers for Green Deal	Lack of end user engagement in assessment, planning and delivery	End-user or change in behaviour undermines retrofit measures and decreases energy reductions anticipated	Building Environmental	–	Gill et al., 2010, Hendrickson, 2010 **B,C,D,E,F**
103	Current Guidance and Regulation	Tenant insistence on energy efficiency improvements	Changes in landlord tenant relationships	Contract termination, homelessness or upgrade and improvements. Changes in GHG emissions	Building Environmental Physical health	+/-	Crewe, 2007 DECC, 2012 **B,C,D,E,F**
104	Current Guidance and Regulation	Increase in fuel costs	Control/ influence of heating/ ventilation behaviour	Reduction of GHG emissions. Positive behaviour change	Environmental	+	Marmot, 2011 **B,C,D,E,F**
105	Current Guidance and Regulation	Increase in fuel costs	Influence of heating/ ventilation behaviour	Increase in fuel poverty and social inequalities between the rich and poor	Environmental Physical health	–	Tod et al., 2012 **C,D,E,F**
106	Current Guidance and Regulation	Increase in fuel costs	Influence of heating/ ventilation behaviour	Reduced ventilation, Increase in indoor pollutant concentrations,	Physical health Building	–	Korjenic et al., 2012 **C,D,E,F**
107	Current Guidance and Regulation	Increase in fuel costs	Influence of heating/ ventilation behaviour	Heat or eat. Reduction in calorific intake to stay warm, consequent health impacts	Physical health Building	–	Tod et al., 2012 **C,D,E,F**
108	Current Guidance and Regulation	Increase in fuel costs	Influence of heating/ ventilation behaviour	Sense of lack of control of life, depression, mental health issues	Building Environmental	–	Thomson, 2013 **C,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
109	Current Guidance and Regulation	Lack of clear construction guidance or holistic investigation of dwelling	Poor or inappropriate application of energy efficiency measures	Negative health impacts, failure to achieve energy saving, damage to fabric and contents. Potential legal challenges.	Building Environmental	–	STBA, 2012 **D,E,F**
110	Current Guidance and Regulation	Need for new designs, equipment, materials and specification	Increase in specialist designers and manufacturers	Economic growth, Growth of UK based manufacturers, supply chains, specialist designers and contractors. Jobs	Building Environmental Physical health	+/-	Santarius, 2012 **B,C,D,E,F**
111	Changes to Design/ Construction and manufacturing Processes	Need for new designs, equipment, materials and specification	Increase in UK based specialist designers and manufacturers	Economic growth, concurrent increase in greenhouse emissions from manufacturing and construction sectors. Failure to achieve GHG emission targets	Economic Environmental	+/-	Santarius, 2012 **B,C,D,E,F**
112	Changes to Design/ Construction and manufacturing Processes	Need for new designs, equipment, materials and specification	Increase in UK based sustainable specialist designers and manufacturers	Environmental impacts. Green Growth	Environmental Psychological Well Being Economic	+	Santarius, 2012 **B,C,D,E,F**
113	Changes to Design/ Construction and manufacturing Processes	Need for new designs, equipment, materials and specification	Increase/ No increase in specialist designers and manufacturers	Limited product selection, compatibility issues, incorrect method statements, wrong priorities, defective detailing. Opposite scenario	Building Environmental Physical health	+/-	Binswinger, 2001 **C,D,E,F**

No	Policy Impact on Buildings			Impacts on People/ Nature		+/-	Reference
				Unintended Consequence	Domain		
115	Changes to Design/ Construction and manufacturing Processes	Need for contractors with specialist knowledge	Increase in skill set of existing workforce, retraining, employment	Reduction in unemployment figures. Increased business opportunities at local, national and international level	Economic Psychological Well Being	+	Marmot, 2012 **C,D,E,F**
116	Changes to Design/ Construction and manufacturing Processes	Need for more contractors with specialist knowledge	Lack of new schemes/training provided. Current workforce used.	Failure to achieve building specification, failure to reach energy targets due to faulty construction going unchecked, commissioning issues	Building Environmental Physical health	–	Pan and Garmston, 2012; Sinnott and Dyer, 2012 **B,C,D,E,F**
117	Changes to Design/ Construction and manufacturing Processes	Need for partnerships between wider group of players	Culture changes/ less adversarial contracting	Productive, efficient and cost effective contracts. Less stressful working relationships	Economic Psychological Well Being	–	Latham, 1994; Egan, 1998 **B,C,D,E,F**
118	Changes to Design/ Construction and manufacturing Processes	Greater coordination of trades	Clear design goals, good project management	Clear and open client/ contractor relationships. Common aim. Achievement of GHG reduction targets	Economic Psychological Well Being	–	Latham, 1994; Egan, 1998 **B,C,D,E,F**
119	Changes to Design/ Construction and manufacturing Processes	Faults in reporting procedures and lack of training of BCO's	Buildings passed as achieving energy reduction targets.	Failure to reach targets in reality, whilst believing these have been achieved. CO_2 emissions higher than anticipated. Climate change impacts. Current testing too limited (ADL1A) and open to abuse by contractors	Environmental	–	Pan and Garmston, 2012 **B,C,D,E,F**

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Appendix C

Additional Archetypes used in Modelling

Ten geometries were used for analysis of the UK domestic stock. Nine were derived from the LUCID project (Oikonomou et al., 2012). The remaining example, House 7, was taken from the Lancet study (Wilkinson et al., 2009). Geometries are shown in Table 1 We make the assumption that these archetypes are adequate to broadly represent the UK domestic stock based on their occurrence in the English Housing Survey (EHS, 2012). Building geometries in Figures 1-10 and Tables 2-11. The models were produced in CONTAM. Indicative examples of the models, outline plans and full dimensions for each property are shown. Dwelling references refer to their allocation in Oikonomou et al. (2012).

Table 1 Archetypes

Stock models used in this study	
Flat 1	1 bed layout 1
Flat 2	1 bed layout 2
Flat 3	1 bed layout 3
House 1	3 bed terrace
House 2	2 bed terrace
House 3	2 bed semi
House 4	5 bed terrace
House 5	3 bed bungalow
House 6	3 bed terrace above shop
House 7	3 bed detached

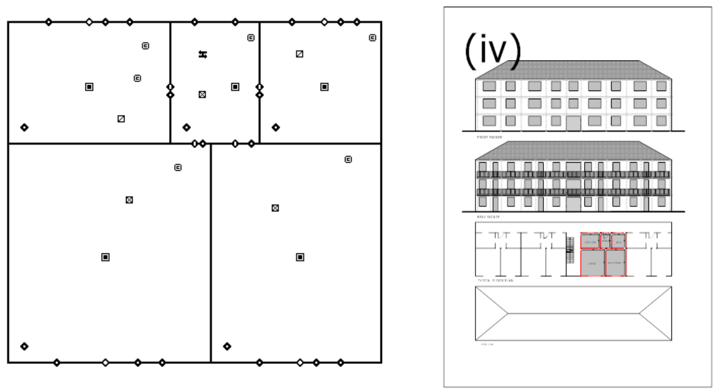


Figure 1 Flat 1: 1 bed (geometric type- IV) dwelling reference H04 & H07

Table2 Flat 1 (1 bedrooms) – dimensions

Flat 1						
Footprint	54.60 m²					
Number of floors	1					
Floor to ceiling height	2.60 m					
Envelope area	180.96 m²					
Permeable envelope	40.56 m²					
Room	Kitchen	Living	Bed 1	Entrance	Bathroom	Total
Floor area m²	8.50	19.35	15.75	4.75	6.25	54.60
Volume m³	22.10	50.31	40.95	12.35	16.25	141.96

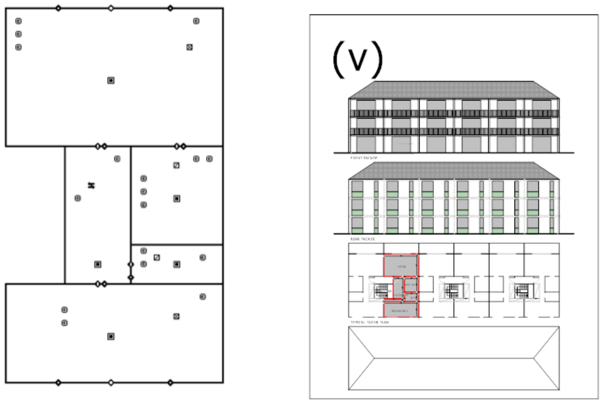


Figure 2 Flat 2: 1 bed (geometric type- V) dwelling reference H06

Table 3 Flat 2 (1 bedroom) – dimensions

Flat 2						
Footprint			50.56 m^2			
Number of floors			1			
Floor to ceiling height			2.70 m			
Envelope area			193.46 m²			
Permeable envelope			39.69 m²			
Room	Kitchen	Living	Bed 1	Entrance	Bathroom	Total
Floor area m^2	5.75	20.16	14.00	7.20	3.45	50.56
Volume m^3	15.52	54.43	37.80	19.44	9.31	136.50

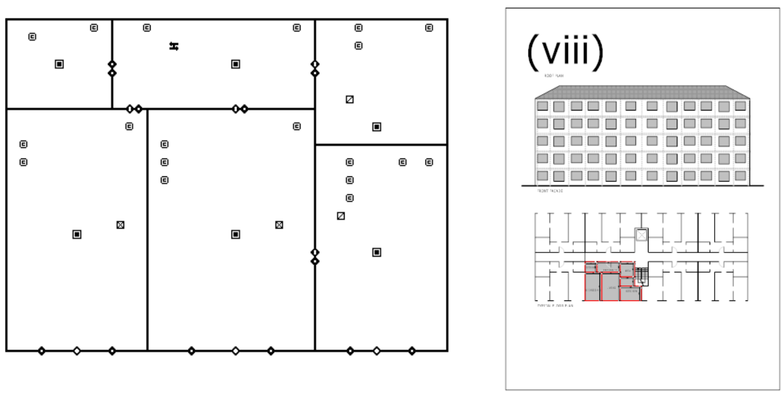


Figure 3 Flat 3: 1 bed (geometric type- VIII) dwelling reference H11, H12 & H15

Table 4 Flat 3 (1 bedroom) – dimensions

Flat 3							
Footprint				52.19 m^2			
Number of floors				1			
Floor to ceiling height				2.60 m			
Envelope area				180.30 m²			
Permeable envelope				21.72 m²			
Room	Kitchen	Living	Bed 1	Entrance	Bathroom	Storage	Total
Floor area m^2	9.43	14.26	12.88	6.44	5.88	3.30	52.19
Volume m^3	24.52	37.08	33.49	16.74	15.29	8.58	135.70

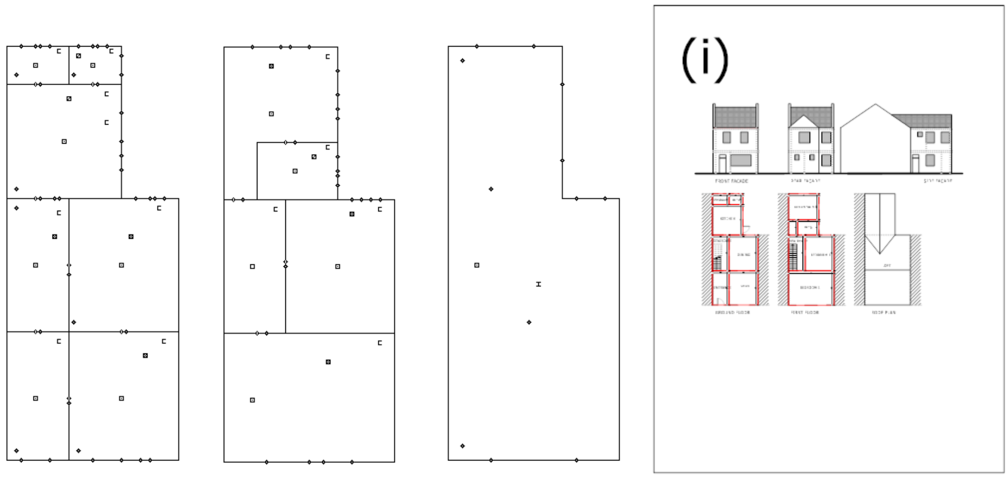


Figure 4 House 1: 3 bed terraced (geometric type- I) dwelling reference H01

Table 5 House 1 Terraced (3 bedrooms) – dimensions

House 1							
Footprint	76.80 m²						
Number of floors	2						
Floor to ceiling height	2.80 m						
Envelope area	476.60 m²						
Permeable envelope	251.04 m²						
Room	Kitchen	Living	Bed 1	Bed 2	Bed 3		
Floor area m²	16.00	17.10	27.00	17.86	16.00		
Volume m³	44.80	47.88	75.60	50.01	44.80		
Storage	Toilet	Hall	Dining	Bathroom	Stair(x 2)	Total	
3.08	2.54	9.90	17.86	5.60	10.33	153.60	
8.62	7.11	27.7	50.08	15.68	28.92	430.12	

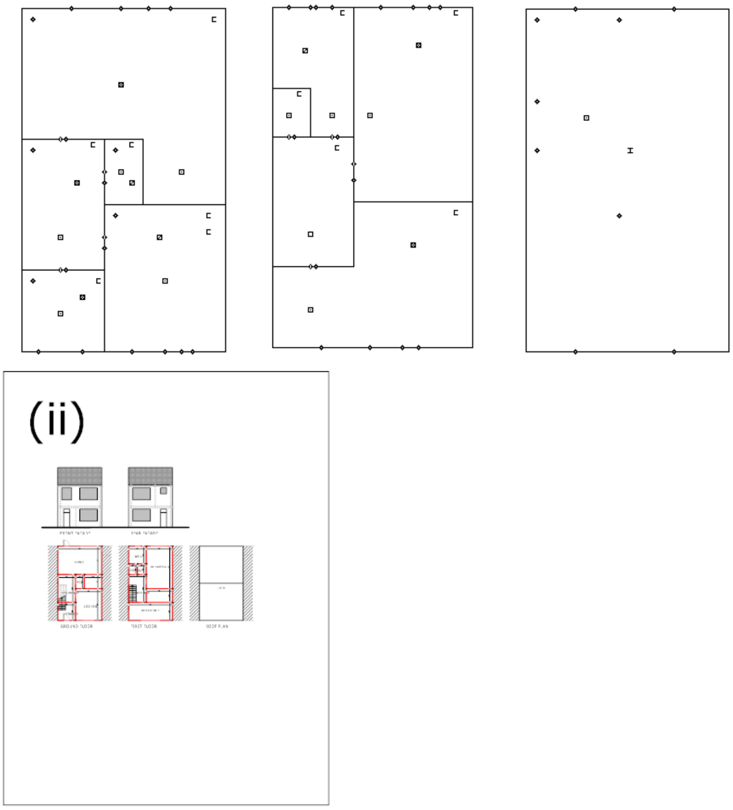
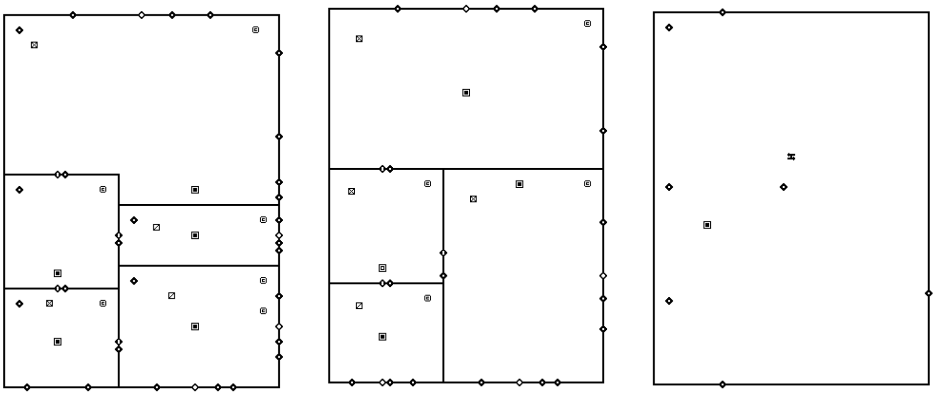


Figure 5 House 2: 2 bed terraced (geometric type- II) dwelling reference H02, H05, H10 &H14

Table 6 House 2 Terraced (2 bedrooms) – dimensions

House 2										
Footprint	65.10 m^2									
Number of floors	2									
Floor to ceiling height	2.80 m									
Envelope area	317.24 m^2									
Permeable envelope	199.64 m^2									
Room	Kitchen	Living	Bed 1	Bed 2	Store	Toilet	Hall	Bath	Stair x2	Total
Floor area m^2	16.66	29.70	23.04	22.20	1.89	2.50	6.38	8.13	9.89	130.20
Volume m^3	46.65	83.16	64.51	62.16	5.29	7.00	17.8	22.76	27.69	364.56



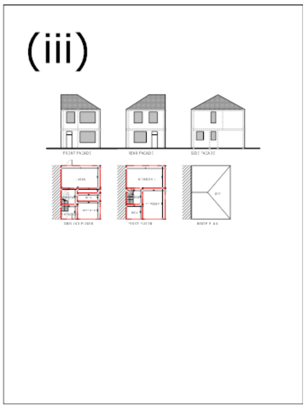


Figure 6 - House 3: semi-detached (geometric type- III) dwelling reference H03

Table 6 - House 3 Semi-detached (2 bedrooms) – dimensions

House 3									
Footprint	49.20 m²								
Number of floors	2								
Floor to ceiling height	2.80 m								
Envelope area	317.24 m²								
Permeable envelope	199.64 m²								
Room	Kitchen	Living	Bed 1	Bed 2	Toilet	Hall	Bath	Stair (x2)	Total
Floor area m²	9.63	23.32	20.70	16.62	4.37	5.64	5.62	6.25	98.40
Volume m³	26.96	65.30	57.96	46.54	12.24	15.79	15.74	17.50	275.52

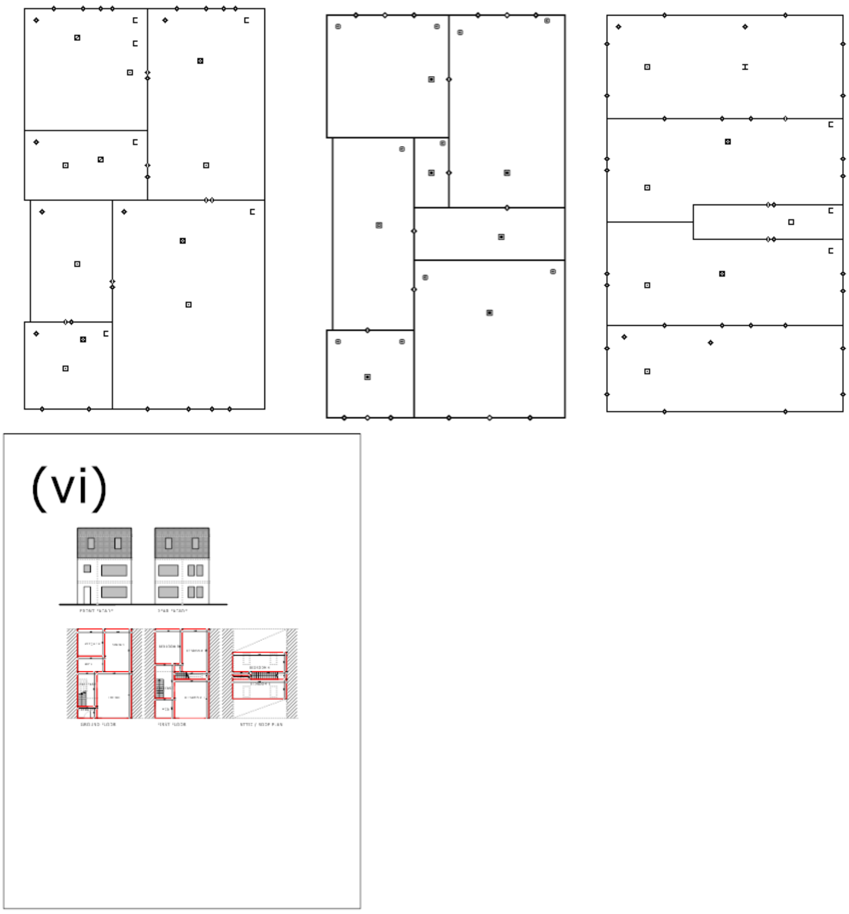


Figure 7 - House 4: 5 bed terraced (geometric type-VI) dwelling reference H08

Table 7 - House 4 Terraced (5 bedrooms) – dimensions

House 4								
Footprint				79.93 m^2				
Number of floors				3				
Floor to ceiling height				2.80 m				
Envelope area				461.95 m^2				
Permeable envelope				331.15 m^2				
Room	Kitchen	Living	Bed 1	Bed 2	Bed 3	Bed 4	Bed 5	Dining
Floor area m^2	11.25	26.70	26.70	18.98	12.25	18.63	18.63	18.98
Volume m^3	31.50	74.76	74.76	53.13	34.30	52.15	52.15	53.13
Room	Dining	Store	Toilet	Total	Bath		Stair x2	Stair x2
Floor area m^2	16.00	17.10	27.00	153.60	6.38		11.25	4.45
Volume m^3	44.80	47.88	75.60	430.12	17.85		31.50	12.46

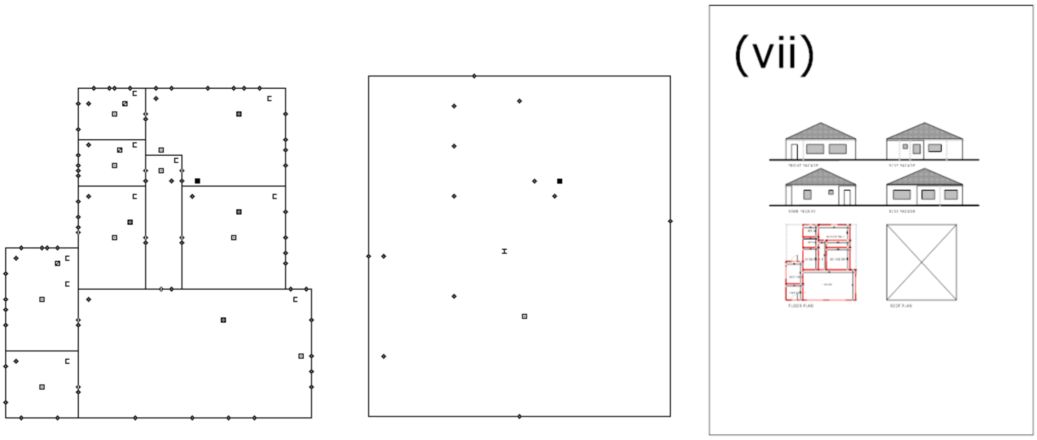


Figure 8 - House 5: 3 bed bungalow (geometric type-VII) dwelling reference H07

Table 8 - House 5 Bungalow (3 bedrooms) – dimensions

House 5										
Footprint	88.66 m²									
Number of floors	1									
Floor to ceiling height	2.80 m									
Envelope area	476.60 m²									
Permeable envelope	251.04 m²									
Room	Kitchen	Living	Bed 1	Bed 2	Bed 3	Entrance	Toilet	Bath	Com	Total
Floor area m²	7.68	32.20	13.50	10.79	7.05	5.52	3.30	3.86	4.76	88.66
Volume m³	21.50	90.16	37.80	30.18	19.75	15.56	9.24	10.78	13.30	248.27

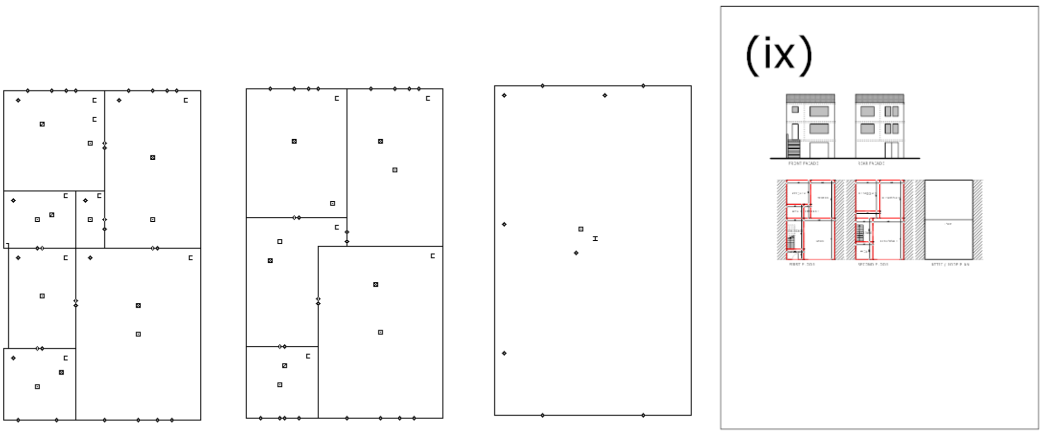


Figure 9 - House 6: 3bed terraced above a shop (geometric type-IX) dwelling reference H13

Table 9 - House 6 Terraced –above shop (3 bedrooms) – dimensions

House 6						
Footprint	79.93 m²					
Number of floors	2					
Floor to ceiling height	2.80 m					
Envelope area	366.50 m²					
Permeable envelope	237.70 m²					
Room	Kitchen	Living	Bed 1	Bed 2	Bed 3	Dining
Floor area m²	12.25	26.70	26.70	18.97	15.75	19.97
Vol m³	34.30	74.76	74.76	53.12	44.10	55.92

Store	Toilet	Hall	Bath	Stair x2	Total
2.00	5.00	3.75	6.37	11.25	159.96
5.60	14.00	10.5	17.84	31.50	447.58

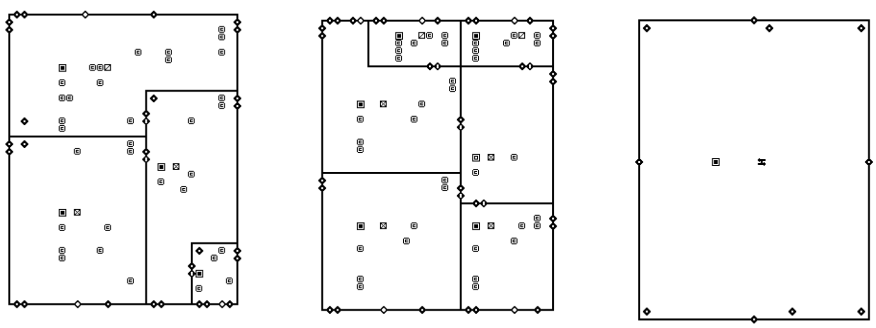


Figure 10 - House 7: 3 bed detached (Lancet: house model)

Table 10 - House 7 Detached house (3 bedrooms) – dimensions

House 7											
Footprint	48.00 m²										
Number of floors	2										
Floor to ceiling height	2.40m										
Envelope area	230.4 m²										
Permeable envelope	230.4 m²										
Room	Kit	Living	Bed 1	Bed 2	Bed 3	Land	Toilet	Hall	En-suite	Bath	Total
Floor area m²	18.60	16.20	12.60	12.60	6.00	9.60	1.80	11.40	3.60	3.60	96.00
Volume m³	44.60	38.90	30.30	30.20	14.40	23.00	4.30	27.40	8.60	8.60	230.40

Appendix D

Calculation of Annual Average London (GLA) Background $PM_{2.5}$ Concentration

The following appendix shows the method of calculation for the annual average urban background $PM_{2.5}$ for 2010 and the production of annual hourly $PM_{2.5}$ files for 2010 and 2050 used in the modelling. Available data downloaded from the Monitors and averaged is shown in Table 1, with the average values by location type shown in Table 2.

Table 1 Annual mean $PM_{2.5}$ levels for the GLA stations with sufficient datasets (>94%) 2010

Site	Reference	Station Category[1]	Network	Annual Mean $PM_{2.5}$ μgm^{-3}
Greenwich Mill Valley	GN2	UB	AURN LAQN	13.68
Camden Bloomsbury	BLO	UB	AURN	14.65
Hackney Clapton	HR1	UB	LAQN	13.05
Harrow Stanmore	HK4	UB	AURN LAQN	11.69
Kensington & Chelsea	KC1	UB	AURN LAQN	13.15
Redbridge	RB3	K	AURN LAQN	20.94
Marylebone Road	MY1	K	AURN	11.70
Brent IKEA	BT4	R	LAQN	15.21
Ealing Acton	EA2	R	LAQN	12.63
Greenwich Plumstead	GN3	R	LAQN	14.33
Greenwich Woolwich	GR8	R	LAQN	18.55
Greenwich Westhorne	GR9	R	LAQN	15.52
Hackney Old Street	HK6	R	LAQN	14.28
Tower Hamlets	TH4	R	LAQN	19.03
Bexley Slade Green	BX1	S	AURN LAQN	13.77*
Bexley Belvedere	BX2	S	LAQN	9.96
Bexley Thamesmead	BX3	S	LAQN	9.74
Bexley SG FDMS	BX9	S	AURN	13.70^
Greenwich Eltham	GR4	S	AURN LAQN	17.54
Richmond NPL	TD0	S	AURN	13.21

(*Source:* Composed from data from the LAQN and AURN networks in the GLA, 2010)

[1] Monitoring sites are divided into different classes depending on their proximity to major pollution sources. Each class of site will be broadly representative of similar locations nearby. Urban background stations will record pollution levels that are similar to those found in nearby residential areas, whereas roadside stations will show levels similar to those obtained from roads of comparable size and traffic flow (LAQN, 2010). Station categories used in this study include: UB= urban background; K= kerbside; R= roadside and S= suburban.

* It is acknowledged that there are known inaccuracies in $PM_{2.5}$ data obtained using TEOM measurement and their lack of equivalence to the European Air Quality Directive (2008/50/EC), even when gravimetric equivalent calculations are applied. However, this is currently the main data available. For this purpose, this data was compared to the few FDMS monitors available to test for accuracy in this situation.

^ Data from FDMS monitoring which also collects the volatile element of $PM_{2.5,}$ show similar values to TEOM measurements and are therefore considered valid for the purpose of this study. This gives an

Average annual mean $PM_{2.5}$ reading for each location type and worst case scenario enabling model parameterisation.

Table 2 LAQN / AURN Network Monitoring Stations with Full Years (>75%) $PM_{2.5}$ Data Sets (2010)

Station Type	n =	2010 Annual Mean $PM_{2.5}$ μgm^{-3}
K	2	16
R	7	15
S	6	13
UB	5	13

(*Source*: Compiled using data from AURN and LAQN Networks, 2010)

Table 3 Urban background $PM_{2.5}$ values in μgm^{-3} From DEFRA Publications* and measured values from AURN/ LAQN (2010)

DEFRA	2007*	2050*
UB	17-20	9
Measured	**2010**	
UB	11 -16	

(**Source*: Williams, 2007)

In order to provide input into the OSPM software (section 4.1.6) for the locational and spatial element of the project; a year's full hourly urban background $PM_{2.5}$ data file was needed for both current and future scenarios. These were to be used in addition to the yearly weather files, in order to create a basis for the pollutant mapping element of the study in OSPM. To produce these files, empirical data from the available GLA based monitors was required. Wilkinson *et al*. (2009) use an average urban background mean yearly $PM_{2.5}$ concentration of 13μgm^{-3} for London, which is confirmed by the authors research on the LAQN and AURN and identifies Hackney Clapton Urban Background (UB) Monitoring Station as having the nearest yearly average to this figure, (13.05 μgm^{-3}) and also approximates to the mean of the UB stations in London that have full data sets for 2010. The data from this station has been fully ratified and is therefore more reliable. A multiplying factor was added to adjust the file to 13.00 μgm^{-3} and to 9 μgm^{-3} for the 2050 file. Downloaded data was formatted and normalised with negative values removed and data gaps infilled with reference to the figures either side of the missing values. Of a total of 8761 possible readings, 327 fell into this category, representing > 0.04% and this was deemed an acceptable level of maximum possible error.

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Wilkinson, P., Smith, K.R., Davies, M., Adair, H., Armstrong, B.G., Barrett, M., Bruce, N., Haines, A., Hamilton, I., Oreszczyn, T., Ridley, I., Tonne, C. and Chalabi, Z. (2009). *Public health benefits of strategies to reduce greenhouse-gas emissions: household energy*. The Lancet, 374, **9705**, 1917-1929

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Appendix E

Uncertainty within the Models and Assumptions

Uncertainties in the Models and Assumptions

The function of this appendix is to elaborate the processes used to quantify the uncertainty of the various simulation outputs caused by the uncertainty of the simulation input variables and to point to future work. This builds on the details of differential sensitivity analysis carried out in chapter 5 sections 5.1.5; 5.1.7 and 5.2.3.

One set of uncertainties relate to potential programming and data errors in the model structure. Extensive testing and use of the modelling has been carried out. It is however possible with the complexities of the modelling that residual errors may occur. However, it is suggested that the extensive use of the models has eliminated these.

Two types of sensitivity can be evaluated: Individual sensitivities, which describe the influence on predictions of variations in each individual input and total sensitivities, which are due to the uncertainties in all the input data. To identify the inputs to which the outputs are particularly sensitive and those to which they are insensitive. It is therefore possible to identify the parameters which must be chosen with care, so that the accuracy of program predictions is not compromised, and the parameters for which accurate specification is unnecessary.

- The first step is to identify key building features to which a particular output, is particularly sensitive in terms of the concentrations of $PM_{2.5}$. This can help guide the designer towards an improved design and the fabricator towards improved quality control in critical areas such as ventilation components in an energy efficiency retrofit.
- Secondly, to set and use default values for some elements of the programme. That is identify parameters which should be removed from the control of the program user because they cannot (except perhaps by very skilled users) be assigned sufficiently accurate values.
- Ascertain the total uncertainty in outputs due to all the input uncertainties. The resolution of the programs (i.e., the maximum accuracy) which can be expected in absolute predictions. This information is important for empirical validation studies, in which predictions are compared with measured data, since it allows sound judgement to be made about the validity of the programs.

Range of Uncertainties

Modelling analyses such as this study rely on multiple assumptions and many uncertainties. Its results should therefore be interpreted only as indicative and relative rather than as precise calculations of impact. Uncertainties in the HIDEEM/SCRIBE tools have been examined by Hamilton et al. (2015). However, there is currently limited observed data on the impacts of retrofitting strategies on indoor air quality and health to compare against CONTAM and EnergyPlus model outputs. Nonetheless, despite these uncertainties, the results can provide important indications of likely impacts that can be used to inform policy decisions.

There are also many uncertainties in the parameters and quantitative relationships on which all the calculations are based. It is impossible to provide a detailed survey of all aspects of such uncertainty, but the sections below list each of the key components that feed in to the impact calculation and the uncertainty associated with each.

Consideration of the model inputs and the effect on outputs reside in several components within the model; broadly, these are: (i) indoor pollutant models (CONTAM/EnergyPlus), (ii) indoor temperature relationship, (iii) housing ventilation mapping, (iv) EHS derived SAP variables, and (v) exposures health impact. This appendix also considers $PM_{2.5}$ variables in particular due to their relevance to this thesis,

Indoor Temperature

The prediction of temperature using the Warm Front (WF) relationship suggests that the EHS is the same as the WF stock. Although there are certainly differences between the WF stock and the EHS there have been several studies comparing this group to the EHS and other UK level sample statistics in looking at both health effects and physical building characteristics (Hamilton et al 2009, Hong et al 2004, Oreszczyn et al 2006). These works showed that the physical building characteristics were broadly similar in the WF and English housing stock and thus offered a sufficient basis from which to use the WF internal temperature and building fabric and heat system efficiency relationship. The differences are likely to lie in various socio-economic indicators that will influence the affordability of warmth.

Ventilation Mapping

The allocation of the dwellings into the ventilation group (i.e. no Extract or Trickle ventilation, Trickle Vents only, Extract Fans only and Trickle Vents and Extract Fans) will have a degree of uncertainty with respect to the limitation some dwellings will have different ventilation systems than allocated in the mapping. The primary reason for the uncertainty is the lack of details of ventilation characteristics in UK houses. We rely on several sources to determine the ventilation system within any given EHS dwelling. The explicit source of ventilation system detailed in the EHS is the presence and operation of Extract Fans, which are thus the most certain system in terms of allocation. The presence of Trickle Vents is based simply on the age of the property, with all new (post 1990) buildings assumed to have Trickle Vents according to the changes in Part F of the building regulations. A further 5% of pre-1990 dwellings are randomly selected from the EHS to account for older dwellings with Trickle Vents. This estimate is based on further analysis of Warm Front to examine the proportion of dwellings with at least 8 Trickle Vents (a value selected on the basis of sufficient presence in the dwelling). The combination of Trickle and Extract will be determined on the presence of Extract Fans and whether the dwelling is new or part of the 5% randomly selected and therefore has Trickle Vents. All other types are judged to have no ventilation system aside from window opening. The matching is based on a hierarchy of dwelling type, floor level (flats only), and size. The exposure levels are generated from the CONTAM model results for a particular dwelling and ventilation type predicted using the SAP modelling permeability.

EHS Derived SAP Variables

The methodology used here is virtually the same as that used in the 2011 DECC housing energy fact file. The assumptions here divided in to two parts (i) EHS Conversion and (ii) SAP prediction. The EHS conversion is detailed further in Addendum A '*Converting EHS variables for use in energy models*' (DECC, 2012). The SAP methodology also has a range of limitations and uses many best estimates to predict indoor environmental features. In both cases there are very likely room for improvements and any given value will be related to the criterion used to generate or match dwelling characteristics and assumptions in the SAP methodology.

Exposure Health Impact

The exposure-response relationships for mortality are evidence-based and were obtained from published epidemiological studies. However, there are two main assumptions which are theoretical (i.e. not strictly evidence-based) and which may have significant impact on the results. The first is the set of disease-specific time functions which were used to account for disease onset and cessation lags over time. The second is that the exposure-response impacts are assumed to be independent.

$PM_{2.5}$: Uncertainties in Inputs into CONTAM/EnergyPlus Models

For the purpose of this investigation, it is assumed that the CONTAM/EnergyPlus modelling as used both individually, and in the case of CONTAM for the HiDEEM/ SCRIBE exposures is accurate from a building physics perspective and computationally precise, with any uncertainties lying in the input values and the post-processing of results. It is acknowledged that this is a big supposition. CONTAM/EnergyPlus have certain accepted in-built modelling assumptions such as the homogenous mixing of airborne pollutants which may not reflect the actual individual exposure experienced and which are difficult to quantify. Computational inaccuracies can occur due to the set-up of simulation parameters; however, these have been previously thoroughly investigated and any issues discovered resolved to ensure accuracy even if their effect was relatively small. An example is the reduction of time-step recording from an hour to a 10 second calculation with a 15-minute output (See figure1).

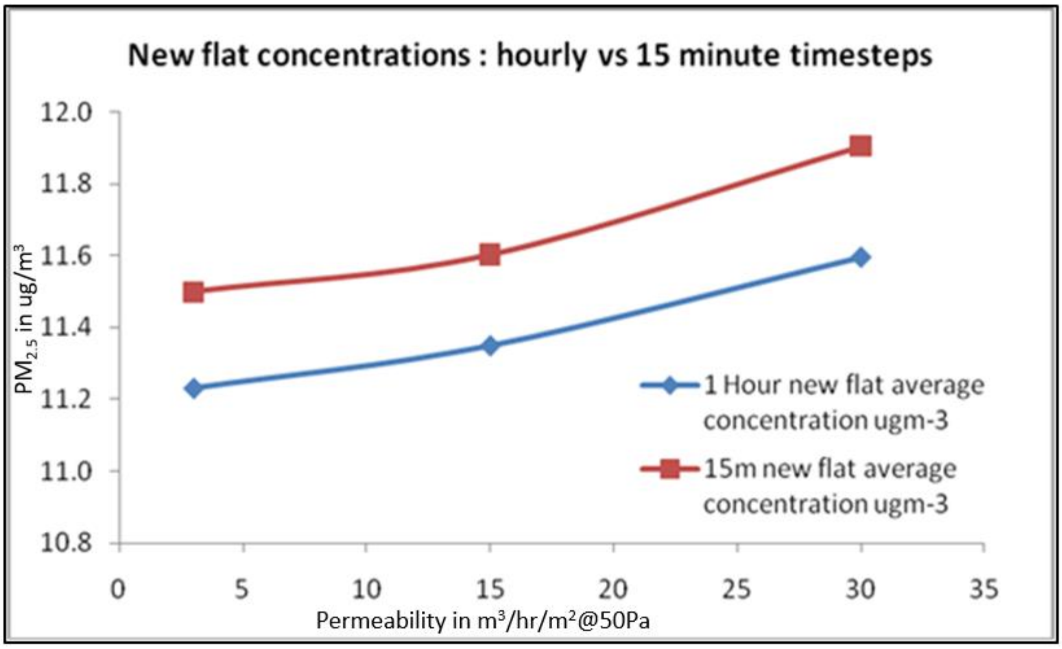

Figure 1 Comparison of $PM_{2.5}$ concentrations (45/10/45 weighted) using 15 minute and one-hour time steps for permeabilities of 3, 15 and 30 m^2/m^3/hr@ 50Pa in a flat.

Permeability in m^3/m^2/hr @50 pa	3	15	30
% difference in readings	2.3	2.2	2.6

The CONTAM manual, (Walton and Dols, 2006) does not deal with the issue of sensitivity analysis or uncertainty in the programme outputs. There is however a number of case studies available on the NIST website at http://www.bfrl.nist.gov/IAQanalysis/case%20studies/cwcase_references.htm#ten

Some uncertainties occurring in model outputs compared to empirical research have also been highlighted by model users and are seen in specific studies e.g. Johnson et al., 2012 (Table 1), with further studies listed under section 3. However, these are not generally validations of exposure outputs directly and tend to focus on ventilation characteristics and energy usage. Their impact on the pollutant outputs of the HiDEEM tool is difficult to quantify without running specific models for each input discovered. It is possible that these could have major, little or no impact at all or be confounded by other inputs. All inputs into the CONTAM modelling are the subject of investigations currently being carried out by Dr Ben Jones (Nottingham), Dr Jon Taylor (UCL) and myself. It is a major exercise and the full results will not be available for some time yet.

Table 1 Examples of Contam outputs and measured data sources

Item	Source	Variations in values	Impacts on predicted pollutant concentrations
Natural ventilation rates	Johnson et al., 2012	Comparison with measurements from literature show that CONTAM generally predicts natural ventilation rates within $\pm$35%	Currently unknown
Wind driven air volume flow rate	Johnson et al., 2012	CONTAM under predicts wind driven air volume flow rate by approximately 25%	Currently unknown

Comparisons: $PM_{2.5}$ Modelled and Measured Data

The main focus of this section is the accuracy of the pollutant outputs in relation to monitored data sources and latest research and how the current inputs used in the modelling relate this data in order to consider uncertainty. Data sources for the main pollutants modelled in HiDEEM, variations in values and its impact on the current pollutant outputs are shown in Table 2

Table 2 Main $PM_{2.5}$ literature and monitoring sources available and their impacts

Item	Source	Variations in values	Impacts on predicted pollutant concentrations
Indoor concentrations of $PM_{2.5}$	Ozkaynak et al., 1996.	Single source: cooking The emission rate of $PM_{2.5}$ from cooking is 1.6 $mg.min^{-1}$ $\pm$ 0.6 $mg.min^{-1}$ based on 4.1 $mg.min^{-1}$ $\pm$ 1.6 $mg.min^{-1}$ of inhalable PM_{10} of which 40% is the finer fraction of $PM_{2.5}$.	Variation in indoor component of $PM_{2.5}$ of ($\pm$ 38%) Taken into account in sensitivity analysis (Shrubsole et al., 2012)
	Shrubsole et al., 2012	Modelled indoor residential concentration for London (with single cooking source) 25 ugm^{-3} $\pm$16 in non-smoking homes.	Sensitivity analysis shows variation of $\pm$66% in overall exposure In addition, once multiple sources are added (Shrubsole et al., 2012) Mean concentration for London increases to 28 $\pm$ 20 ugm^{-3} ($\pm$72%) and ($\pm$107%) for smoking households. Based on sensitivity analysis of various key components.
	Hanninen et al., 2004;	European cities Athens 23 $\pm$ 11 ugm^{-3}; Basle 17$\pm$ 8 ugm^{-3}; Prague 25 $\pm$ 16 ugm^{-3} and Helsinki 25 $\pm$ 16 ugm^{-3}. (Hanninen et al. 2004).	
	Lai et al., 2004	Oxford UK, mean residential indoor $PM_{2.5}$ concentration 17.3$\mu g.m^{-3}$	
	Wallace et al., 2006	US study mean indoor (non-smoking) concentrations of 25.8$\mu g.m^{-3}$ with a range of 7.2-66.0$\mu g.m^{-3}$	

Item	Source	Variations in values	Impacts on predicted pollutant concentrations
External $PM_{2.5}$ concentrations from monitoring stations	AURN, 2012.	A single value (13 ugm^{-3}) representing the urban background (2010) and (9 ugm^{-3}) for 2050 is used in modelling for chapter 5. This is varied in the subsequent chapters depending on location. Variation in background levels occur based on location. Changes to the indoor $PM_{2.5}$ concentrations from the external $PM_{2.5}$ component is based on building envelope permeability.	A range of 10-16 ugm^{-3} is seen based on (2012 data) i.e. current value $\pm$ 23%. On average this would result in a maximum increase in overall indoor $PM_{2.5}$ concentrations of $\pm$ 12 % based on an average I/O ratio of 0.5
External $PM_{2.5}$ concentrations from monitoring stations	Kings, 2012 (Tim Baker)	Errors in ratified data from monitoring stations as shown (unpublished source)	This would add an additional average error of $\pm$ 2 ugm^{-3} ($\pm$13-20%) to the external concentrations.
External $PM_{2.5}$ penetration factor	Chen and Zhao, 2011	A penetration factor of used has been used for all infiltration pathways in HiDEEM, OK for window opening, but overestimation for other infiltration pathways. Range of values 1-0.6 in the literature	Minimal impact, except on external $PM_{2.5}$ components in high permeability buildings with little window opening Covered in sensitivity analysis (chapter 5)

The key issue here is whether to further research individual uncertainties for each exposure type, or to amalgamate them to give an overall uncertainty. If it is the latter, then a method needs to be investigated/developed that takes into account the fact that some exposures have greater levels of uncertainty than others.

Literature Sources

Table 3 is divided into (A) those papers validating CONTAM; (B) modelling studies that have some link to monitored data (but not necessarily in relation to HiDEEM/SCRIBE modelled exposures) and (C) literature that explores uncertainty in relation to our current modelled pollutant exposures.

Table 3 Literature sources used

A. CONTAM Validation Papers
• Haghighat F. and Megri A.C., 1996. A Comprehensive Validation of Two Airflow Models - COMIS and CONTAM. *Indoor Air*, pp.278–288. • Emmerich, Steven J, 2001. Validation of Multizone IAQ Modeling of Residential-Scale Buildings: A Review. *ASHRAE Transactions*.

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B. Modelling Studies Using CONTAM with Some Comparison to Monitored Data

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- Further examples are available at http://www.bfrl.nist.gov/IAQanalysis/case%20studies/cwcase_references.htm#ten

C. Literature Data Sources Uncertainty Review of Current Modelled Pollutant Exposures

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Regional Average Annual $PM_{2.5}$ External Concentrations

Regarding the fixed annual external $PM_{2.5}$ concentrations currently used in modelling, there are two issues that require attention:

- Whether this value is appropriate for other regions/locations in England other than London and other location types (e.g. urban vs rural) and what if any errors exist in the data?
- Whether a fixed value rather than a constructed variable hourly file is appropriate?

For the first point, the UK Air Quality Strategy published in July 2007, for the first time included the proposed EU move from PM_{10} limit values to a $PM_{2.5}$ limit values. As a result, data for most of the UK is only available for a few years. However, outline trends can be seen and if a current value is to be used that is not projected into the future, use of these figures seems appropriate. The Automatic Urban and Rural Network (AURN) operate a range of monitors in the UK of which approximately 57 currently capture $PM_{2.5}$ in England. These sites are classified according their location type: BU= Background Urban, BR= Background Rural, TU= Traffic Urban, IU= Industrial Urban, BS= Background Suburban. These sites have been investigated to obtain annual average concentrations for 2011 and 2012 as seen in table 4 below and allocated to the government regions as seen in both HiDEEM and the SCRIBE tool.

Table 4: AURN monitoring site and annual average $PM_{2.5}$ concentrations for 2011 and 2012

Site	Type	Region	Reg. No	Annual Average Concentration $PM_{2.5}$ µgm-3	
				2011	2012
Rochester Stoke	BR	S East	8	14	14
London Bexley	BS	London	7	15	12
London Eltham	BS	London	7	16	13
Birmingham Acocks	BU	W Midlands	5	None available	11
Birmingham Tyburn	BU	W Midlands	5	16	14
Bristol St Paul's	BU	S West	9	15	13
Chesterfield	BU	E Midlands	4	14	12
Eastbourne	BU	S East	8	16	16
Harwell	BU	S East	8	12	13
Hull Freetown	BU	Yorkshire	3	12	11
Leeds Centre	BU	Yorkshire	3	16	16
Leicester Centre	BU	E Midlands	4	14	14
Liverpool Speke	BU	N West	2	12	11
London Bloomsbury	BU	London	7	17	16
London Harrow	BU	London	7	16	12
London N. Kensington	BU	London	7	16	15
London Teddington	BU	London	7	17	14
Manchester Piccadilly	BU	N West	2	14	14
Middlesbrough	BU	N East	1	11	10
Newcastle Centre	BU	N East	1	12	10
Norwich Lakenfields	BU	E of England	6	14	14
Nottingham Centre	BU	E Midlands	5	13	12
Oxford St Ebbes	BU	S East	8	12	12
Plymouth Centre	BU	S West	9	11	N/A
Portsmouth	BU	S East	8	16	14
Preston	BU	N West	2	11	11
Reading New Town	BU	S East	9	14	12
Sheffield Centre	BU	Yorkshire	3	17	16
Southampton Centre	BU	S East	9	16	15
Stoke-on-Trent Centre	BU	W Midlands	5	16	15
Sunderland Silksworth	BU	N East	1	15	N/A

Site	Type	Region	Reg. No	Annual Average Concentration PM2.5 µgm-3	
				2011	2012
Warrington	BU	N West	2	13	13
Wirral Tranmere	BU	N West	2	10	10
York Bootham	BU	Yorkshire	3	N/A	10
London Harlington	IU	London	7	16	13
Salford Eccles	IU	N West	2	16	13
Birmingham Tyburn	TU	W Midlands	5	17	13
Camden Kerbside	TU	London	7	16	13
Chatham Roadside	TU	S East	8	17	17
Chepstow A48	TU	S West	8	17	12
Chesterfield Roadside	TU	E Midlands	4	14	15
Haringey Roadside	TU	London	7	N/A	18
Leeds Headingley	TU	Yorkshire	3	19	17
London Marylebone	TU	London	7	24	21
Stanford-le-Hope Roadside	TU	E England	6	18	15
Stockton-on-Tees Eaglescliffe	TU	N East	1	12	11
Storrington Roadside	TU	S East	8	16	16
York Fishergate	TU	Yorkshire	3	N/A	13

There is insufficient rural monitoring on the network, with the only background rural (BR) station in Rochester Kent operating. The geographical location of this station, capturing as it does pollution down-wind from London is not likely to be indicative of rural levels elsewhere. It therefore seems inappropriate for use in other locations which are likely to be lower in concentration. There is also insufficient Background Suburban (BS) data and this is based in two London boroughs. The same applies to industrial urban (IU) although this category of location is rare. Background Urban (BR) values are usually taken to be indicative of the general level of pollution in a borough and by implication region. As can be seen, there are not huge variations for this type of monitoring station around England, although there are indicative levels for many major cities. For traffic urban (TU) the range is broader and there is a further differentiation between kerbside and roadside. Mean values for each station type and possible ranges are shown in table 5.

Table 5 Analysis of monitoring station data by type

Monitor type	Mean value		Median		Mode		Range of values	
	2011	2012	2011	2012	2011	2012	2011	2012
BU= Background Urban	14.1	13.0	14.0	13.0	16.0	13.0	10.0-17.0	10.0-16.0
BR= Background Rural	14.0	14.0	14.0	14.0	14.0	14.0		

TU= Traffic Urban	17.0	15.1	17.0	15.0	17.0	13.0	12.0-24.0	11.0-21.0
IU= Industrial Urban	16.0	13.0	16.0	13.0	16.0	13.0		
BS= Background Suburban	15.5	12.5	15.5	12.5	15.5	12.5		

The range seen in table 5 could provide a possible input for sensitivity analysis where data for a particular Background Urban setting is not available. Urban background $PM_{2.5}$ concentrations for individual regions are shown in Table 6.

Table 6 Regional mean urban background $PM_{2.5}$ concentrations

Region	Mean value µgm-3
1. North East	10.0
2. North West	11.8
3. Yorkshire and the Humber	13.3
4. East Midlands	13.0
5. West Midlands	13.0
6. East of England	14.0
7. London*	13.0
8. South East	13.5
9. South West	13.3

*London figures are based on additional data from the GLA network.

Even though the monitored data has been ratified, errors occur. Tim Baker of Kings College London kindly sent me a table of (unpublished) uncertainties in the values of monitored data (Table 7) part of which is reproduced below. We should take this into account in any further sensitivity analysis.

Table 7 Uncertainties in the London Air Quality Network (LAQN) monitoring data

Concentration in $\mu g.m^{-3}$	uncertainty ± in $\mu g.m^{-3}$	% uncertainty ±
0	1.9	190
5	1.9	38.0
10	2	20.0
15	2	13.0
20	2	10.0
25	2.1	8.4
30	2.2	7.3
35	2.3	6.6

The may be other local authority/borough stations, which could provide in theory further data sources if required, but access to these is not easily available. An alternative is to use the DEFRA 1 km square mapping for specific cities, although for regions (if the data is available), however, this is a very large and time consuming exercise. As regards fixed external vs hourly value files in Contam/Energy Plus, these have been previously constructed by the author and they are relatively straight forward. Findings were that there was very little difference when considering yearly exposure for health calculations, although the distribution of values during the year would be very different. Again, this could be a matter for further sensitivity analysis using the different archetypes and ventilation strategies.

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Appendix F

Details of Excel Macro ‘CONTAM-Batch’

This appendix gives details of the macro created to run CONTAM models in batch mode to save excessive computing time. It was developed in conjunction with Dr Phillip Biddulph, UCL. The pcontam program is designed to run CONTAM batch jobs. An archetype template (project) prj file is made using the CONTAMW and saved. Pcontam can then be used to create and run different variations of the template prj file. A series of post simulation commands are available to decode and collate the output simulation files as ntuple files (Excel spreadsheets) with hourly $PM_{2.5}$ concentrations for each room. These can be post-processed to give annual averages or combined and run with a further macro to investigate the individual exposures of occupants as they move around the building.

Running pcontam

Pcontam consists of two files pcontam.BAT, which is just the DOS command wrapper and pcontam.pl, which does all the work. Both files are placed same directory as the template prj file. The instructions and data for Pcontam for each batch job are held in a normal script txt file. To run a set of instructions held in the script file "sample.txt", use the command window and move to the correct directory and type at the command line; > Pcontam.BAT sample.txt

In the script file blank lines and lines beginning with a "!" are ignored. Pcontam reads the file from the top. Each command has two parts separated by a blank space. The first part is the command and the second is the data to be used with that command. Both command and data are case sensitive. If a parameter is set, then it will be used for all subsequent prj file updates.

Script File Commands

Command	Data	Comment
wthfile	The absolute directory path and file name of the weather file.	Replaces the template weather file. Use 'No_change' to use the weather file as defined in the template.
t_list	The timestep for contamx to produce data to the sim file	Shorter timesteps will lead to slower running.
t_scrn	The timestep for contamx to write a progress report to the screen.	Should be as long as possible to speed up contamx.
date_1	The end date of the simulation.	
afmult	'airflow element','multiply factor'	The multiplying factor to the specified airflow element. If the airflow element is 'ALL', all airflow elements will be multiplied by factor. Individual factors override always the 'ALL' command.
afoff	'airflow element','offset factor'	The offset factor to the specified airflow element. If the airflow element is 'ALL', all airflow elements will be offset by factor. Individual factors override always the 'ALL' command.
run	prj file to be run by contamx	Runs contam x using the prj file.
sse	'source name','factor'	Multiply the source strength by factor.
ccdef	'species name','factor'	Multiply the default concentration of species by factor.
updateprj	template prj','new prj'	The new prj file is created using the template file. If the new prj file already exists it is overwritten.
postsim	prj file	The sim file created from the prj file is decoded and a new ntuple file is created for each zone with the time evolving concentration of contaminants for each zone in the project.
mergedata	prefix,file_1.prj,file2.prj …	This command expects that the 'postsim' has been run on all the files in the list. The individual ntuple files are reread and the data from the different projects for each zone are merged into a larger file, one for each zone and with the "prefix" at the start of the new file.
mergespec	prefix,file_1.prj,file2.prj …	This command also expects that the 'postsim' has been run on all the files in the list. The individual ntuple files are reread and the data from the different projects for each zone are merged into a larger file, one for each zone and with the "prefix" at the start of the new file. The prefix is also used to specify which contaminant is listed from each of the prj files.

Appendix G

Additional Research Carried Out Using the Thesis Models

Introduction

The models were constructed such that a range of pollutants sources from both outdoor and indoor sources could be added and the outputs run simultaneously with those for $PM_{2.5}$. The function of this section is to the illustrate the versatility and wider applications of the stock models created to investigate the questions raised within this thesis, but used within other research and contexts outside the scope of this study.

The PURGE Project

The Public health impacts in urban environments of greenhouse gas emissions reduction strategies (PURGE) project, funded by the European Union seventh framework programme FP7/2007-2013 under grant agreement No 265325, saught to investigate the urban impacts of greenhouse gas emissions reduction strategies. One of its foci was the housing sector and the examination of the impacts of policies to reduce uncontrolled ventilation. Aditionally, much of the work and background to chapter 6 is contained in report 'Summary for Policy Makers and Dissemination Guidelines for the EU' (PURGE, 2013, 2014) for which the author with UCL colleagues was a main contributor for the UK stock examples (London and Milton Keynes) and responsible) the author researched all the data on both locations to produce the stock profiles, energy eficiency and ventilation measures and pollutant inputs (including PM2.5) and was also responsible to produce the documents as first author. The investigation in chapter 6, comparing locations was funded by this project. In addition, a study on the impacts of airtightness and various ventilation scenarios on radon concentrations was conducted as follows:

Home Energy Efficiency and Radon

While control of ventilation is good for energy efficiency and reducing end use energy demand, improving indoor temperatures in winter and preventing the high ingress of outdoor pollutants particularly $PM_{2.5}$ (Hänninen et al., 2005), it has the potential to increase concentrations of pollutants arising from sources inside or underneath the home (Nazaroff, 2013). Notable among these is radon, a naturally occurring inert gas formed from the radioactive decay of elements of the uranium series, which seeps into homes through the floor, especially in areas with predisposing geology and soil type (Miles et al., 2007). Radon is the second most important risk factor for lung cancer after smoking and may be responsible for around 1400 cases annually in the United Kingdom (Darby et al., 2005). Radon is unique in the context of IAQ since it is a continuous source, which is therefore not responsive to the intermittent ventilation techniques that can be used to deal with other pollutants at the emission source, for in example using extraction fans to remove cooking related $PM_{2.5}$.

The study considered a range of future energy efficiency scenarios applied to the current English housing stock. The author carried out all the building-related modelling and post-processing of $PM_{2.5}$

concentrations and contributed as second author in the published research paper (Milner et al, 2014), which was the first building modelling study ever published by the British Medical Journal. Health impact assessments were carried out by staff at LHSTM. See Appendix G for the full paper.

This study found that energy efficiency interventions that increase the air tightness of dwellings without compensatory PPV will increase indoor radon concentrations and associated lung cancer risks, a ssimlilar conclusion to that seen in chapter 6, when comparing the stocks of London and Milton Keynes. Specifically, that the reduced air changes associated with energy efficiency upgrades to meet CO_2 reduction targets were likely to increase radon levels by over 50% in the general UK population with an additional annual health burden of close to 5000 life years lost from lung cancer. The addition of PPV partially removed this burden, but at the loss of enrgy efficiency gains. Aside from the widespread use of mechanical ventilation and heat recovery (MVHR), ventilation related improvements (e.g. extract fans, trickle vents etc.) with energy efficiency interventions can be achieved only at the expense of additional radon related lung cancer burdens unless there is widespread use of remediation.

The HIDEEM Project

The Health Impact of Domestic Energy Efficiency Measures (HIDEEM) model has been developed for The Department of Energy and Climate Change (DECC) by the UCL Energy Institute and the Complex Built Environment Systems Group of the UCL Institute for Environmental Design and Engineering (IEDE), in collaboration with the London School of Hygiene and Tropical Medicine (LSHTM).

The UK housing stock is expected to undergo a transformation in terms of energy efficiency, initiated by a variety of government programmes. These will have different direct impacts on human health while addressing carbon emissions and alleviating fuel poverty. The mix of impacts on both costs to government and benefits to human health need to be reflected in on going impact and sustainability assessments. The aim of the HIDEEM model is provide estimates of indoor environmental exposures experienced in the GB housing stock and changes in exposures following the application of a variety of energy efficiency measures of the type and scale detailed in DECC's broad-ranging programme of interventions, and any resulting change in health.

The model broadly includes two main components:

- Building physics-based models of the indoor environment in UK houses (including: temperature, concentrations of particle pollution, second hand tobacco smoke, radon, and risk of mould growth) to which the author contributed by providing the CONTAM models developed for this thesis, and
- Models to quantify associated health impacts of exposure changes using life table methods.

Using the HIDEEM tool to calculate the value of health benefits of installing solid wall insulation in all properties in England, DECC calculated that this would give a total improvement in people’s health of between £3.5-£5 billion over the lifetime of the measures. DECC’s modelling work using HIDEEM also suggests there are substantial health-related costs associated with cold homes, and DECC’s Fuel

Poverty Framework (DECC, 2013) notes that 'for this reason, we should continue to prioritise vulnerable poor households for support'. The model continues to be used and updated.

The research has generated a number of publications to which the author contributed where indoor concentrations of $PM_{2.5}$ in the domestic stock were investigated and/or explained in detail: Hamiliton et al. (2012) Model documentation for the HIDEEM tool where the author provided all the pollutant modelling protocols and data; Hamilton et al. (2015) a peer reveiwed publication (see appendix H), UCL (2104a) an evaluation of CONTAM models used in the development of the HIDEEM model. A CONTAM tutorial and design exercise has been prepared, to which the author contributed UCL (2014b). In addition, the author has presented at numerous conferences and events as an invited speaker and been an author in numerous peer reviewed publication (see introduction).

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Appendix H

Selected Peer-Reviewed Publications

Atmospheric Environment 62 (2012) 336–343

Contents lists available at SciVerse ScienceDirect

Atmospheric Environment

journal homepage: www.elsevier.com/locate/atmosenv

Indoor $PM_{2.5}$ exposure in London's domestic stock: Modelling current and future exposures following energy efficient refurbishment

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H I G H L I G H T S

► We model the current (2010) and future (2050) domestic stock for London.
► We examine the effects of energy efficiency measures on indoor $PM_{2.5}$ concentrations.
► Decreases in permeability combined with MVHR systems substantially reduce exposure.
► Occupant behaviour plays a critical role in determining $PM_{2.5}$ exposure level.

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A B S T R A C T

Simulations using CONTAM (a validated multi-zone indoor air quality (IAQ) model) are employed to predict indoor exposure to $PM_{2.5}$ in London dwellings in both the present day housing stock and the same stock following energy efficient refurbishments to meet greenhouse gas emissions reduction targets for 2050. We modelled interventions that would contribute to the achievement of these targets by reducing the permeability of the dwellings to 3 $m^3\ m^{-2}\ h^{-1}$ at 50 Pa, combined with the introduction of mechanical ventilation and heat recovery (MVHR) systems. It is assumed that the current mean outdoor $PM_{2.5}$ concentration of 13 μg m^{-3} decreased to 9 μg m^{-3} by 2050 due to emission control policies. Our primary finding was that installation of (assumed perfectly functioning) MVHR systems with permeability reduction are associated with appreciable reductions in $PM_{2.5}$ exposure in both smoking and non-smoking dwellings. Modelling of the future scenario for non-smoking dwellings show a reduction in annual average indoor exposure to $PM_{2.5}$ of 18.8 μg m^{-3} (from 28.4 to 9.6 μg m^{-3}) for a typical household member. Also of interest is that a larger reduction of 42.6 μg m^{-3} (from 60.5 to 17.9 μg m^{-3}) was shown for members exposed primarily to cooking-related particle emissions in the kitchen (cooks). Reductions in envelope permeability without mechanical ventilation produced increases in indoor $PM_{2.5}$ concentrations; 5.4 μg m^{-3} for typical household members and 9.8 μg m^{-3} for cooks. These estimates of changes in $PM_{2.5}$ exposure are sensitive to assumptions about occupant behaviour, ventilation system usage and the distributions of input variables (±72% for non-smoking and ±107% in smoking residences). However, if realised, they would result in significant health benefits.

1. Introduction

Motivated in large part by the desire to pursue CO_2 reduction targets for mitigating climate change, the energy efficiency of new and existing buildings in the UK is likely to be substantially improved over the coming decades (HM Government, 2010a), with existing dwellings projected to account for approximately 80% of the housing stock in 2050 (Boardman, 2008). To meet 2050 greenhouse gas reduction targets, current proposals suggest that

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these existing dwellings should undergo extensive retrofitting, with the installation of insulation and more efficient heating systems coupled with an increase in air tightness (Wilkinson et al., 2009). However, such changes to air tightness and ventilation are likely to lead to changes in indoor air quality (IAQ) and personal exposure to airborne pollutants such as particulate matter (PM) (Milner et al., 2005), the smaller fractions of which are particularly harmful to health (COMEAP, 2009). As most people in developed countries typically spend more than 80% of their time in indoor environments (Klepeis et al., 2001), changes in domestic IAQ consequent to energy efficiency measures may impact on population health (Wilkinson et al., 2009).

Concentrations of $PM_{2.5}$ in houses are affected by the infiltration of outdoor particles, emissions from indoor sources and the removal from the internal air by deposition, filtration and exfiltration, though some re-suspension also occurs largely related to domestic activities (Gehin et al., 2008). In apartments (as well as terraced and semi-detached houses), inter-dwelling transfer of contaminants via party wall permeability is possible (Molnár et al., 2007).

Externally, various factors including building location, height, orientation to outdoor pollutant source and meteorology affect outdoor $PM_{2.5}$ contributions to indoor concentrations (Godish and Spengler, 2004; Patra et al., 2008). In the future, these external concentrations of $PM_{2.5}$ are expected to decline due to reductions in transport emissions and gaseous precursors which produce secondary particles (Williams, 2007). Increased air tightness and installation of mechanical ventilation and heat recovery systems (MVHR) which filter out $PM_{2.5}$, could reduce the penetration of externally generated $PM_{2.5}$ into dwellings. However, any increase in air-tightness without an increase in controlled purpose provided ventilation could lead to a rise in exposure from internally generated $PM_{2.5}$ (Wilkinson et al., 2009).

Indoor $PM_{2.5}$ concentrations are also affected by transient emissions from internal sources such as construction materials, fixtures and fittings and appliances as well as intermittent emissions such as the burning of fuels and candles, smoking, cooking, heating and human domestic activities (Milner et al., 2005; Weschler, 2009). Studies have shown high $PM_{2.5}$ indoor concentrations relative to external levels, with cooking and smoking being the two primary sources (Jones et al., 2000). Occupant movement and behaviour, including window opening, can affect indoor concentrations (Andersen et al., 2009). Within dwellings, different rooms could be subject to very different levels of $PM_{2.5}$, depending on the activities conducted in them (Dimitroulopoulou et al., 2006). Methods are needed to assess the impact of cooking, smoking and domestic activities as well as ventilation behaviour in order to fully understand the impact of energy efficient refurbishment on future exposure.

This paper presents modelling evidence of indoor exposure to $PM_{2.5}$ in current London dwellings, and an assessment of the likely impact of energy efficiency measures designed to meet 2050 climate change mitigation objectives.

2. Materials and methods

2.1. Modelling of exposure to $PM_{2.5}$

The study was based on the application of CONTAM (Emmerich, 2001), a validated multi-zone IAQ model, to predict concentrations of particles with maximum aerodynamic diameter of 2.5 microns ($PM_{2.5}$) from both indoor and outdoor sources, in specific zones/rooms of dwellings. This modelling develops previously published methods of exposure characterization to $PM_{2.5}$ (Wilkinson et al., 2009). It includes a detailed approach to modelling the effect of ventilation systems within dwellings, multiple $PM_{2.5}$ sources with occupant behaviour, location and sensitivity analysis. It uses various empirical data sources as model inputs. An outline of the modelling approach is shown in Fig. 1.

The modelling was carried out to simulate indoor $PM_{2.5}$ concentrations in both houses and apartments using dwelling characteristics selected to be broadly representative of the London housing stock. Simulations were run to investigate the influence of combinations of key parameters: dwelling type, geometry, ventilation system and permeability (Table 1). All model scenarios were run with and without a source of tobacco smoke, and with and without other indoor sources of particles so as to quantify the separate contributions of smoking and particles of indoor and outdoor origin to the overall indoor $PM_{2.5}$ concentrations.

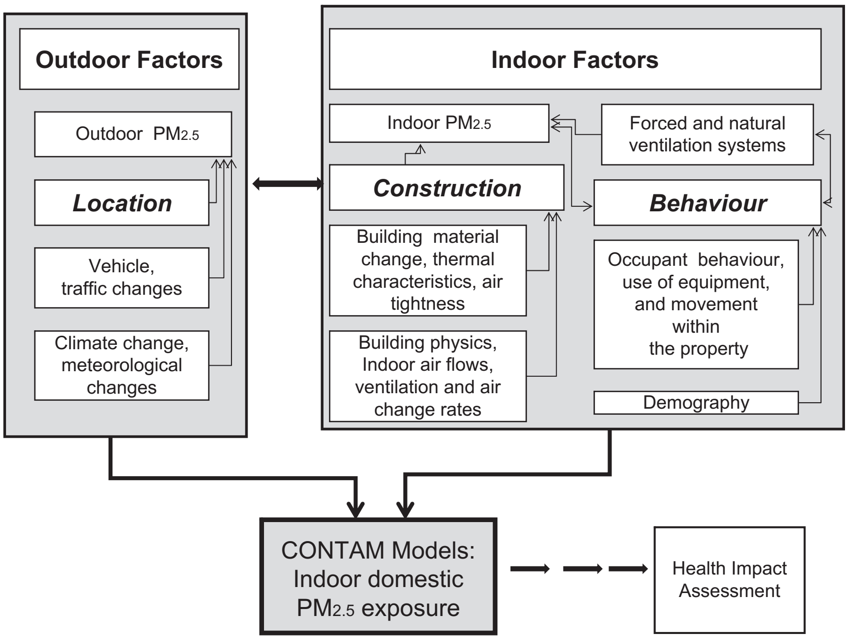

Fig. 1. Range of factors affecting modelling of indoor $PM_{2.5}$ exposure and preparation of data for future input to heath impact assessment.

Table 1
Summary of key dwelling features, $PM_{2.5}$ sources and external environment characteristics used for the specification of baseline simulations of the 2009 and 2050 housing stock.

Common to all simulations of current (2010) and future (2050) stock	Dwelling type	House or apartment[a]	
	$PM_{2.5}$ sources and schedules	Models run (1) with $PM_{2.5}$ source and (2) no source scenario	
	1. Cooking	15 min morning and 30 min evening cooking, with an additional lunch period of 30 min at weekends (gives 1.6 mg min^{-1} emissions of $PM_{2.5}$)	
	2. Smoking	2 cigarettes in the kitchen on weekdays and weekends and 4 cigarettes on weekdays and 7 at weekends in the living room (giving 0.99 mg min^{-1} emissions of $PM_{2.5}$ at 5 min per cigarette)	
	3. Sweeping	Entrance/bathrooms and en-suites on Wednesday and Saturday only 5 min per room (giving 0.05 mg min^{-1} emissions/re-suspension of $PM_{2.5}$	
	4. Vacuuming	All other rooms on Wednesday and Saturdays only, 5 min per room in rotation (giving 0.07 mg min^{-1} emissions/re-suspension of $PM_{2.5}$)	
	5. Dusting	All rooms Saturdays only, 20 min per room in rotation (giving 0.09 mg min^{-1} emissions/re-suspension of $PM_{2.5}$	
	6. Washing machine	In Kitchen, scheduled for 30 min, 3 times a week (giving 0.12 mg min^{-1} emissions of $PM_{2.5}$)	
	7. Washing/showering	Bathroom and En-suite, daily morning and evening schedule for 30 min (giving 0.04 mg min^{-1} emissions of $PM_{2.5}$)	
	Weather	CIBSE/Met Office hourly weather data – Test Reference Year and Design Summer Year	
		Current (2010) housing stock	Stock under future (2050) scenarios
Specific to simulations of 2010 or 2050 scenario	Ventilation regimes	(1) Infiltration and purge ventilation only: (2) Infiltration, trickle ventilators, extraction fans and periodic purge ventilation;	Ventilated via MVHR systems with background /boost modes and filters that remove 80% of $PM_{2.5}$
	Permeability	3, 5, 7, 10, 15, 20, 25, 30 $m^3\ m^{-2}\ h^{-1}$ at 50 Pa	3 $m^{-3}\ m^{-2}\ h^{-1}$ at 50 Pa + MVHR systems with filters removing 80% of $PM_{2.5}$.
	Outdoor $PM_{2.5}$	13 $\mu g\ m^{-3}$	9 $\mu g\ m^{-3}$

[a] The apartment (Fig. 2) was modelled to be on the ground floor, with no adjustments for either changes in wind speed or $PM_{2.5}$ concentrations with height. Emission inventories are based on data from Ozkaynak et al. (1996), He et al. (2004) and Afshari et al. (2005).

2.2. Data inputs and assumptions

The key input parameters to the CONTAM models are summarized in Table 1. The emission rate of $PM_{2.5}$ from cooking was assumed to be 1.6 mg min^{-1} $\pm$ 0.6 mg min^{-1} based on 4.1 mg min^{-1} $\pm$ 1.6 mg min^{-1} of inhalable PM_{10} of which 40% is the finer fraction of $PM_{2.5}$ having a $PM_{2.5}$ deposition rate of 0.39 h^{-1} (figures derived from the large scale PTEAM study (Ozkaynak et al., 1996)). However, published emission rates for cooking vary greatly, depending on food type, cooking method, appliance and method of measurement (He et al., 2004; Olson and Burke, 2006). Consequently, for sensitivity analysis, models were run using estimates to $\pm$2.33 standard deviations. For resuspension from dusting and vacuuming or sweeping, the initial surface loadings reported in Ozkaynak et al. (1996), He et al. (2004) and Afshari et al. (2005) were assumed to be applicable to the London stock.

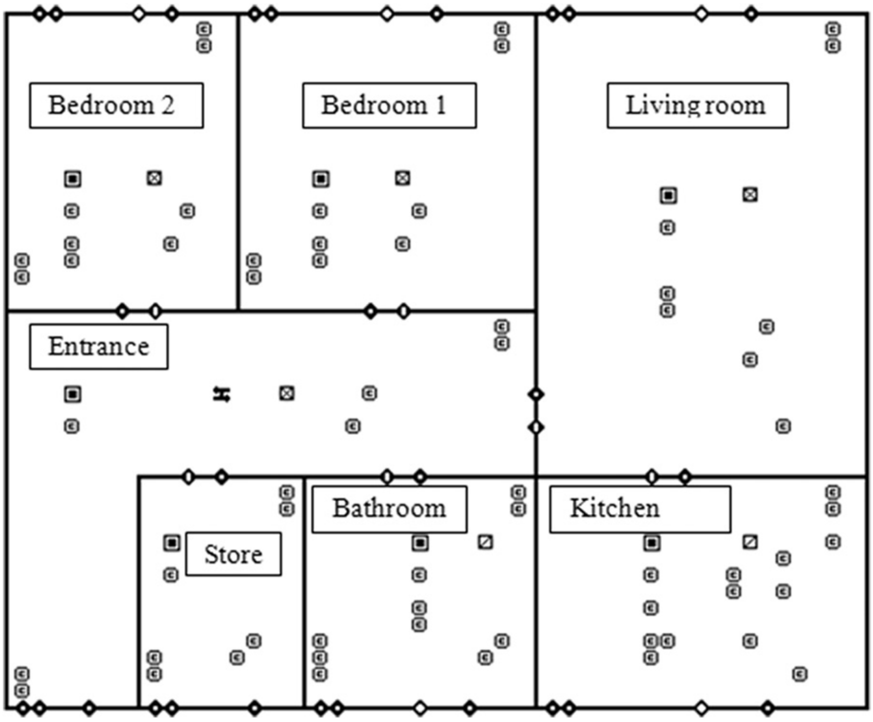

Fig. 2. Example plan of simulated apartment in CONTAM with modelled rooms, pollutant sources/sinks, ventilation systems, windows, doors and adventitious infiltrations (the house was modelled over four levels).

All model simulations assumed that the dwelling permeability is provided by adventitious openings (gaps and cracks) in the external walls, floors and roofs, with gap size proportional to facade area and assuming a crack is situated at the base and top of each wall (Orme and Leksmono, 2002). A penetration factor of 1 was used for all infiltration pathways representing the maximum for $PM_{2.5}$. However, as component size, indoor/outdoor pressure differences, penetration geometry and roughness may affect this factor; sensitivity analysis contrasts this value with 0.6 from Chen and Zhao (2011).

Based on the absence of suitable data, fixed periods and durations for all domestic activities were assumed (Table 1). In dwellings occupied by a smoker, and using data from the Office of National Statistics data (ONS, 2000), it was assumed 1 cigarette was smoked per waking hour; the schedule assumes smoking occurs both outdoors and indoors in the kitchen and living room. Extractor fans, trickle-ventilators and MVHR systems were specified to comply with Approved Document F of the Building Regulations for England (HM Government, 2010b). This document stipulates boost flow rates in kitchens and bathrooms during such events as cooking and washing and background rates at other times to achieve minimum whole building ventilation. Equipment was assumed to be correctly fitted and perfectly functioning.

For the *present day* stock we ran models with two alternative ventilation strategies:

(1) Ventilation achieved via adventitious openings, trickle ventilators, intermittent extract fans and periodic purge ventilation by window opening (representing 20% of stock, which has been refurbished, or constructed in line with current regulations).
(2) Ventilation achieved via adventitious openings and periodic purge ventilation by window opening but without trickle ventilators or extraction fans (80% of stock).

These proportions of the stock were informed by the number of dwellings built post 1990 when amendments to Part F of the Building Regulations (1990) were introduced requiring trickle ventilation and extract fans, and data from the Warm Front study, which estimated the percentage of pre-1990 properties already fitted with trickle vents and assumed to have intermittent extract fans (ONS, 2011; Warm Front, 2011).

Outdoor conditions are a key factor influencing window opening behaviour and also subject to high uncertainty (Andersen et al., 2009). For the London stock, we assumed a seasonal variation where windows were opened to 10% of the maximum aperture for 8 h during the summer months and closed during the winter months (except during purge events for cooking, bathroom and toilet usage) and then subjected these schedules to sensitivity analysis. Eight levels of permeability were used for exterior façades: 3, 5, 7, 10, 15, 20, 25 and 30 $m^3\ m^{-2}\ h^{-1}$ at 50 Pa. These values reflect the observed distribution of the UK domestic stock (Stephen, 1998) and are assumed to be broadly representative of London; being confirmed by later studies showing little change even among some new build properties (Stephen, 2000; Grigg, 2004).

Internal walls are considered impermeable. Doors when shut, have gaps between door and frame. Internal doors (excluding storage) are always open except during activities such as cooking and bathroom use. For 'new' dwellings, a gap beneath the door exists to allow airflow for correct functioning of the MVHR system in line with Approved Document F (2010).

For the outdoor $PM_{2.5}$, the mean annual average concentration from 20 urban background monitoring stations in the Automatic Urban and Rural Network (AURN) for London and the London Air Quality Network (LAQN, 2010) was used. Annual mean $PM_{2.5}$ concentration was 13 $\mu g\ m^{-3}$ with a variance of 2.9 $\mu g\ m^{-3}$. Input weather data were derived from the CIBSE/Met Office hourly data – Test Reference Year (TRY) and Design Summer Year (DSY) and were used for the models runs for current day (2010) conditions. Dynamic indoor temperature profiles were informed by a study from FMNectar (2007), which investigated ventilation effectiveness in support of Part F of the Building Regulations. They have a range of average temperatures between zones of (18.75–20.35 °C) for winter and (23.65–24.35 °C) for summer scenarios.

For the *2050 housing stock*, we assumed that all dwellings were refurbished to a permeability of 3 $m^3\ m^{-2}\ h^{-1}$ at 50 Pa and are ventilated by MVHR systems with filters that remove 80% of $PM_{2.5}$. The choice of this scenario was motivated to meet UK targets which aims to reduce CO_2 emission by 80% by the year 2050 (CCC, 2011). The assumption of a complete building refurbishment to this standard and installation of MVHR to all of the London stock is a deliberately extreme scenario to illustrate the maximum feasible impact on the indoor environment. For outdoor $PM_{2.5}$ a concentration of 9 $\mu g\ m^{-3}$ was assumed (Williams, 2007). To investigate the effects of changing climate on the future (2050) scenario, weather files were created by adjusting current day files in line with climate models based on particular emission scenarios from UK Energy Research Centre (UKERC, 2009) using the method proposed by Belcher et al. (2005). However, as these showed no significant impact (<0.1%) on annual average indoor $PM_{2.5}$ concentrations, weather files representing 2010 were used throughout this study.

2.3. Personal exposure and occupancy schedules

Personal exposure to $PM_{2.5}$ was estimated from the simulations for three categories of occupancy schedule: (a) a 'household average' concentration of $PM_{2.5}$ in the living room, bedroom and kitchen using time weighting factors of 0.45, 0.45, and 0.1 respectively (Wilkinson et al., 2009); (b) the exposure experienced by a 'cook' who occupies the living room, bedroom and kitchen during periods of cooking using weighting factors of 0.56, 0.36 and 0.08 on weekdays and 0.4, 0.5 and 0.1 at weekends respectively; and (c) the exposure of a person who never enters the kitchen and only spends time in the living room and bedroom with weighting 0.62 and 0.38 on weekdays and 0.58 and 0.41 at weekends respectively. Weighting factors for occupancy schedules (b) and (c) were derived by adjusting the time spent in relevant rooms from the central case (a) (Wilkinson et al., 2009). For each category of occupant, the mean indoor $PM_{2.5}$ exposure across the London stock was calculated from the simulations using weightings that reflect the frequency (proportion) of the permeability distribution within the UK domestic stock (Stephen, 1998) and assuming 50% of current London dwellings are apartments, and 50% houses (ONS, 2011).

2.4. Sensitivity analysis of estimates of $PM_{2.5}$ exposure

Differential sensitivity analysis (DSA) was carried out to examine the sensitivity of the results to model inputs and assumptions. This method assumes that the effect of each variable is independent and additive. For numerical parameters (e.g. $PM_{2.5}$ emission and deposition rates) high and low values were calculated as the means ±2.33 standard deviations; the range that encompasses 99% of the values assuming normally distributed data (Lomas and Eppel, 1992). For other variables such as window opening, where field data are sparse, we proposed values to reflect the range of normal behaviour, changing the open period by ±2 h and increasing the aperture to 40% of the total window area. For building orientation, the dwellings were rotated in steps of 45°.

Additional investigations of the effect of location on external $PM_{2.5}$ concentrations within the Greater London authority (GLA) were carried out using the Operational Street Pollution Model (OSPM) (Vardoulakis et al., 2007). Dwellings were classified into three broad exposure categories – high, moderate and low – based on distance from busy streets and/or intersections (Vardoulakis et al., 2008). OSPM was run using composite meteorological and urban background $PM_{2.5}$ files of 13 μg m^{-3} for the present day and 9 μg m^{-3} for 2050. Vehicle traffic and emission data for A (Major road, non-motorway) and minor roads for the same years were constructed with eight typical London street configurations.

Model runs using different weather files (Heathrow TRY, Heathrow DSY and Gatwick International Weather for Energy Calculation (IWEC)) (ASHRAE, 2010), were performed to consider possible effects of locational meteorological variation (CIBSE, 2010). Other analyses examined the effect of altering the height of the infiltration gaps by ±0.1 m from the initial height of 2.3 m. Variation in room volumes was informed by Chapman (1994) based on the range of storey heights (2.3–2.6 m) within the GLA. These yield average room volume changes of ±8.6% from the baseline models.

The maximum and minimum alternative values of each input parameter were entered into the model while holding all other variables constant. The results are reported as the percentage difference in $PM_{2.5}$ concentrations compared with the central baseline estimate. In the case of building orientation the mean deviation from the baseline value (north) was calculated.

3. Results

The results of the CONTAM simulations of $PM_{2.5}$ exposure are presented in Table 2a and b.

3.1. Non-smoking households

The simulations suggest that under present day conditions, average indoor concentrations of $PM_{2.5}$ are appreciably higher than those in the outdoor air because of indoor sources. Thus, in non-smoking dwellings, although indoor levels of $PM_{2.5}$ derived from outdoor air are less than half the outdoor levels, the concentration experienced by the average household member indoors was estimated to be 28.4 μg m^{-3}, over twice the concentration in the outdoor air (13.0 μg m^{-3}). Most of the contribution to this very high level of indoor particle exposure was from cooking-related sources as indicated by the difference in exposure of the cooks and non-cooking occupants.

Under the 2050 refurbishment scenario, household average exposure to total $PM_{2.5}$ (from indoor and outdoor sources) was reduced from 28.4 μg m^{-3} to 9.6 μg m^{-3} (−66%) as a result of permeability reductions and the application of correctly installed and perfectly functioning MVHR equipment. The contribution from external sources represents 23% of the current total indoor $PM_{2.5}$ exposure and 15% in 2050. Average London domestic stock indoor/outdoor (I/O) ratios for $PM_{2.5}$ from external sources are 0.5 for present day and 0.2 for 2050 due to the decrease in stock permeability and filters on the MVHR system.

Separate 2050 scenarios with the proposed reduction in permeability to 3 m^3 m^{-2} h^{-1} at 50 Pa but *without* providing an MVHR system, results in an increase in the London annual average indoor exposure to total $PM_{2.5}$ of 5.4 μg m^{-3} from the baseline of 28.4 μg m^{-3}. For cooks, the increase is 9.8 μg m^{-3} from the baseline of 60.5 μg m^{-3} and for non-cooks, an increase of 1.5 μg m^{-3} from 15.5 μg m^{-3}. These increases are due to the influence of decreases in outdoor $PM_{2.5}$ penetration and reduced ventilation of the $PM_{2.5}$ from indoor sources.

There was considerable variation in $PM_{2.5}$ exposure levels among household members. The simulations show that in non-smoking households peak exposure levels are related to periods of cooking in the kitchen as noted by others (Ozkaynak et al., 1996; Weschler, 2009). The results suggest cooks experience twice the level of $PM_{2.5}$ exposure of the average household member, and more than four times that of a 'non cook' who does not enter the kitchen. This is because the average cook is exposed to 5.8 times the internally generated $PM_{2.5}$ compared with the average non cook, while both are exposed to roughly similar levels of externally generated $PM_{2.5}$. The household average $PM_{2.5}$ exposure (the time-weighted average of $PM_{2.5}$ experienced in the living room, bedroom and kitchen) approximates the average exposure of a family of one cook and three non-cook members (average exposure = 26.8 μg m^{-3}).

3.2. Smoking households

According to the English Housing Survey 2009, the proportion of properties in London with smokers is 18.9% (EHS, 2009). For smoking households, the concentration of $PM_{2.5}$ experienced by the average household member was 57.8 μg m^{-3}, over four times the outdoor concentration. The external $PM_{2.5}$ component now represents a substantially smaller proportion (11%) of the overall exposure compared to the non-smoking scenario. The non-cook receives a similar exposure to the average household member (57.1 μg m^{-3}) due to $PM_{2.5}$ emissions from smoking occurring in the living room. The cook experiences an annual average increase in

Table 2
a) Dwellings without smoking occupants: simulated average annual indoor $PM_{2.5}$ exposures for the present day and 2050. b) Dwellings with smoker occupants: simulated average annual indoor $PM_{2.5}$ exposures for the present day and 2050.

Year	Exposure model	Indoor exposure to $PM_{2.5}$ from indoor sources (row percent)	Indoor exposure to $PM_{2.5}$ from outdoor air (row percent)	Total	Change in total indoor $PM_{2.5}$ 2010–2050
a) Annual average indoor $PM_{2.5}$ (μg m^{-3})					
Present day (external $PM_{2.5}$ 13.0 μg m^{-3})	Household average	22.0 (77%)	6.4 (23%)	28.4	
	Cook	54.2 (90%)	6.3 (10%)	60.5	
	Non-cook	9.4 (61%)	6.1 (39%)	15.5	
2050 (External $PM_{2.5}$ 9.0 μg m^{-3})	Household average	8.2 (85%)	1.4 (15%)	9.6	−66%
	Cook	16.5 (92%)	1.4 (8%)	17.9	−70%
	Non-cook	3.1 (70%)	1.3 (30%)	4.4	−71%
b) Annual average $PM_{2.5}$ (μg m^{-3})					
Present day (external $PM_{2.5}$ 13.0 μg m^{-3})	Household average	51.4 (89%)	6.4 (11%)	57.8	
	Cook	96.0 (94%)	6.3 (6%)	102.3	
	Non-cook	51.0 (95%)	6.1 (5%)	57.1	
2050 (external $PM_{2.5}$ 9.0 μg m^{-3})	Household average	26.5 (95%)	1.4 (5%)	27.9	−52%
	Cook	46.1 (97%)	1.4 (3%)	47.5	−54%
	Non-cook	32.8 (96%)	1.3 (4%)	34.1	−40%

$PM_{2.5}$ exposure of +41.8 μg m^{-3} compared to the non-smoking scenario (60.5–102.3 μg m^{-3}). The 2050 refurbishments reduce the indoor exposure substantially. However, with reduced permeability and the MVHR system designed according to Approved Document F (HM Government, 2010b) occupants still experience exposures between 3.1 and 5.3 times the external $PM_{2.5}$ concentration of 9.0 μg m^{-3}.

3.3. Sensitivity analysis

The results of the sensitivity analysis are shown in Table 3. The percentage change in $PM_{2.5}$ concentration for each variable represents its independent effect on the baseline model results. The quadrate sum is calculated from the square of these values, enabling the overall error in the baseline values to be obtained (Lomas and Eppel, 1992).

Indoor $PM_{2.5}$ deposition and emission rate and window opening behaviour had the largest influence on the overall $PM_{2.5}$ concentrations, with generally smaller impacts from building orientation, infiltration height, volume, indoor temperature and external weather conditions. Results from OSPM confirmed location as a less critical variable as they showed a maximum $PM_{2.5}$ external variation of 2.22 μg m^{-3} against an urban background value of 13.0 μg m^{-3}, with a variation of 1.38 μg m^{-3} against a 2050 urban background value of 9.0 μg m^{-3} in the 2050 scenario. The quadrate sum for smoking (±106.7%) and non-smoking properties (±72.4%) showed very large variations in $PM_{2.5}$ exposure are possible. It should be stressed that there is variation in the uncertainty of the individual parameters used. For example, variability in building height and volume (Chapman, 1994) is better understood than variability in behaviour and window opening (Andersen et al., 2009).

4. Discussion

4.1. General findings

This study provides new insights into the potential effect of changes to the energy efficiency of London's housing stock on exposure to $PM_{2.5}$, and the potential for the reduction of other airborne pollutants by the use of extraction fans and filtration of pollutants through mechanical ventilation systems. The results suggest that domestic energy efficiency interventions of the type and scale needed to meet 2050 climate change abatement objectives could yield substantial net reductions in $PM_{2.5}$ exposure. However, the magnitude and directions of exposure changes depend on the details of the specific mitigation measures, and adverse effects may occur, for example, if air-tightness is achieved without the associated installation and maintenance of correctly functioning ventilation systems. The results for both smoking and non-smoking households represent only time spent in the indoor domestic environment. In order to quantify overall personal exposure to $PM_{2.5}$ and consequent health impacts, time spent in other microenvironments (e.g. in transport, at work) and outdoors will need to be taken into account (Wallace et al., 2006).

The use of MVHR systems could result in a substantial reduction of indoor sources (including tobacco smoke), and markedly lower exposures from outdoor sources. For the exposures seen, the simulated changes are likely to be positive for health, which may add to the case for pursuing energy efficiency interventions – if properly implemented. The results should be interpreted with caution as they are dependent on the range of assumptions and input parameters specified. In particular, field studies show high variability in $PM_{2.5}$ emission rates from cooking: 2.4 ± 2.1 mg min^{-1} (He et al., 2004), 36 ± 98 mg min^{-1} (Olson and Burke, 2006), 1.6 ± 0.6 mg min^{-1} (Ozkaynak et al., 1996). Similarly, there are variations in deposition rate calculation methodologies with differing interpretations of surface area (Fogh et al., 1997; Thornburg et al., 2001), which could lead to differences in absolute $PM_{2.5}$ exposures.

The sensitivity analysis indicates that there are large uncertainties in $PM_{2.5}$ emissions and deposition rates which influence exposure. The other major factors affecting personal indoor exposure appear to relate to changes to the building envelope, ventilation systems and occupant activity.

The specification of the 2050 stock was deliberately based on an extreme scenario, with all dwellings reduced in permeability to 3.0 m^3 m^{-2} h^{-1} and fitted with MVHR systems, combined with effective 80% particle filtration. However, the UK's current 'Retrofit for the Future' programme contains examples of construction refurbishment projects in London employing MVHR systems and achieving substantial reductions in permeability, in some cases to the Passivhaus standard of 0.6 m^3 m^{-2} h^{-1} (LEB, 2011). So, whilst such permeabilities are achievable, the impacts seen on $PM_{2.5}$ levels are therefore likely to be towards the maximum of what could be achieved. They do however, suggest possible positive effects on $PM_{2.5}$ exposures from energy efficiency measures implemented as part of a strategy for meeting abatement targets as specified by the UK Climate Change Committee (CCC, 2011). To make clear the effect of these housing changes, the models assumed no changes in behaviour or new technologies, which could further influence indoor air quality, save for the assumption of a lower outdoor $PM_{2.5}$ concentration in 2050.

4.2. Comparison with empirical studies

The results of this study are broadly consistent with on-site measurements of average annual indoor domestic $PM_{2.5}$ concentrations. Hanninen et al. (2004) as part of the EXPOLIS project monitored indoor $PM_{2.5}$ concentrations in non-smoking households in four European cities (some of which may represent an appropriate comparison to London), showing the following variations: Athens 23 ± 11 μg m^{-3}; Basle 17 ± 8 μg m^{-3}; Prague 25 ± 16 μg m^{-3} and Helsinki 25 ± 16 μg m^{-3}. Our estimated non-smoking household average exposure for London of 28.4 μg m^{-3} is slightly higher. Wallace et al. (2006) monitoring 36 residences in North Carolina over a year show mean indoor $PM_{2.5}$ concentrations of 25.8 μg m^{-3} with a range of 7.2–66.0 μg m^{-3} for non-smoking

Table 3
Effect of assuming high and low values for key input parameters in model simulations. Figures indicate the percentage changes in average annual indoor personal $PM_{2.5}$ exposure when each of the listed parameters took high or low values with all other input parameters held constant. All models were for present day London stock based on the 45%, 45%, 10% occupation scenario.

Variable	Average % difference in $PM_{2.5}$ estimate
Building orientation	±3.7
Infiltration height	±0.2
Volume	±4.2
Indoor temperature	±0.6
Window opening	±18.3
$PM_{2.5}$ infiltration rate	±9.0
$PM_{2.5}$ Emission Rate	±36.2
$PM_{2.5}$ deposition rate	±59.1
Weather file changes	±1.3
Quadrate sum	±72.4[a]

[a] In smoking houses this rises to ±106.7% from the baseline rate based on the increase in $PM_{2.5}$ emissions.

households. Substantial variations are seen in empirical studies on smoking concentrations including $PM_{2.5}$. For smoking properties a consumption of 7.4 cigarettes per day results in an average indoor $PM_{2.5}$ concentration of 132.7 $\mu g\ m^{-3}$ measured over 14 days while 4 cigarettes over 19 days yields 66.0 $\mu g\ m^{-3}$ (Wallace et al., 2006). The present study shows a $PM_{2.5}$ concentration of 57.8 $\mu g\ m^{-3}$ for 7 cigarettes per day modelled over a year. Dimitroulopoulou et al. (2005) in a monitoring study on kitchens in 37 new homes in the UK with smokers found 24 h mean $PM_{2.5}$ concentrations of 113 $\mu g\ m^{-3}$ in winter and 134 $\mu g\ m^{-3}$ in summer compared to an annual average of 102 $\mu g\ m^{-3}$ in this study.

4.3. Comparison with other modelling studies

The results of this modelling study are generally consistent with those of other published research. Dimitroulopoulou et al. (2006), using the INDAIR probabilistic model calculated annual indoor mean $PM_{2.5}$ concentrations of 19.78 $\mu g\ m^{-3}$ (calculated to correspond to the base case (a) occupancy scenario) in households with gas cooking, with a peak value of 318 $\mu g\ m^{-3}$ (compared with 442 $\mu g\ m^{-3}$ for this present study) and a standard deviation of 78 $\mu g\ m^{-3}$ in the kitchen. Fabian et al. (2012), using CONTAM to model low-income multifamily housing with a higher cooking emission rate of 1.56 mg min^{-1} calculated a mean indoor $PM_{2.5}$ concentration of 52.9 $\mu g\ m^{-3}$ with a standard deviation of 41.4 $\mu g\ m^{-3}$. Emmerich et al. (2005), using CONTAM modelling as part of the U.S. Dept. of Housing and Urban Development's Healthy Homes Initiative, found the two most effective intervention strategies for indoor air quality were extract fans, if operated during source events (kitchen fan airflow rate 47 l s^{-1}) and efficient air filtration on heating ventilation and air conditioning (HVAC) systems, if operated for a minimum 15% of the time. Our result for the household average $PM_{2.5}$, I/O ratio for the 2050 scenario of 0.15 for an external concentration of 9 $\mu g\ m^{-3}$ using an MVHR system at 80% filter efficiency are consistent with results from Macintosh et al. (2010) with a $PM_{2.5}$, I/O ratio 0.1 for an external concentration of 15 $\mu g\ m^{-3}$.

Future work with a larger set of geometries that are more representative of the London stock could consider a probabilistic approach such as Monte Carlo Analysis (MCA) which was deemed computationally prohibitive and beyond the scope of this study. Unlike DSA, MCA is unable to detect effects of component variables (Dutton et al., 2008). However, in reality many of the input variables within the CONTAM models are linked, e.g. window opening and air change rates, and as such they are not truly linear and superposable (an assumption of DSA techniques used). MCA could provide an indication of homes with overall characteristics likely to lead to higher indoor concentrations of $PM_{2.5}$ (Dimitroulopoulou et al., 2006).

5. Conclusions and future work

This study has developed and applied a series of model simulations in CONTAM to quantify the changes in indoor domestic exposure to $PM_{2.5}$ in the Greater London Area as a result of the application of energy efficiency measures to meet 2050 greenhouse gas abatement targets. It has quantified the key variables influencing indoor $PM_{2.5}$ exposure and shown that construction and occupant factors are major influences, with building location having a relatively smaller effect. The method used in this paper could be applied to assess a wide variety of refurbishment strategies, differing pollutants and changes to occupant behaviour.

Although there are uncertainties associated with the results, our primary findings suggest substantial reductions in $PM_{2.5}$ exposure which are likely to be beneficial for health in most cases if the interventions are implemented appropriately. The work also confirms that present day high exposures for cooks from particle emissions during cooking in domestic environments are avoidable through a comparatively simple adaptation such as the introduction of extraction equipment as noted by Fabian et al. (2012); or by properly fitted, maintained and operated MVHR systems. These also help to remove indoor $PM_{2.5}$ derived from tobacco smoke, which would be beneficial for non-smokers in such dwellings. Further work is now needed to produce a wider range of geometries based on built-form data for the Greater London Area focusing on variations in $PM_{2.5}$ exposure between building geometries, temperature regimes, occupant behaviour, particle penetration and seasonal/diurnal changes in external $PM_{2.5}$ concentrations.

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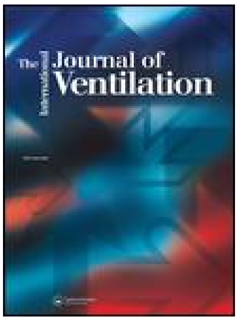

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The Effect of Party Wall Permeability on Estimations of Infiltration from Air Leakage

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The Effect of Party Wall Permeability on Estimations of Infiltration from Air Leakage

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Abstract

The importance of reducing adventitious infiltration in order to save energy is highlighted by the relevant building standards of many countries. This operational infiltration is often inferred via the measurement of the air leakage rate at a pressure differential of 50 Pascals. Some building codes, such as the UK's Standard Assessment Procedure, assume a simple relationship between the air leakage rate and mean infiltration rate during the heating season, the so-called *leakage-infiltration ratio*, which is scaled to account for the physical and environmental properties of a dwelling. The scaling does not take account of the permeability of party walls in conjoined dwellings and so cannot be used to differentiate between the infiltration of unconditioned ambient air that requires heating, and conditioned air from an adjacent dwelling that does not. This article evaluates the leakage infiltration ratio afresh using a theoretical model of adventitious infiltration for a conjoined dwelling. The model is used to predict the mean infiltration rate during the heating season for an apartment and a terraced house located in 14 different UK cities for two extreme assumptions of party wall permeability. The first assumption is that party walls are permeable – this results in a predicted leakage-infiltration ratio that is significantly greater than that used by building codes to evaluate the energy and environmental performance of dwellings. The second assumption is that party walls are impermeable – this results in a predicted leakage-infiltration ratio close to that used by building codes. Knowledge of party wall permeability is not provided by a standard measurement of air leakage but is shown to be vital for making informed decisions on the implementation of energy efficiency measures. These findings have significant energy and health implications and should be of great interest to the policy makers of any country with a large number of conjoined dwellings.

Key words: infiltration, air leakage, permeability, energy, dwelling, apartment, house, multifamily, modelling, CONTAM, AIDA, standard assessment procedure.

1. Introduction

The infiltration of cold air through adventitious openings can be a significant component of a dwelling's heating load. In the UK, for example, this has been recognised by a relevant standard for new dwellings (HM Government, 2010). However, measuring infiltration is technically difficult, invasive and expensive. Accordingly, infiltration is often inferred from a measurement of the Air Leakage Rate (ALR), the rate of airflow through the fabric of a building measured at a steady high pressure difference, normally 50 Pascals (Pa), when the effects of wind and buoyancy forces are effectively eliminated (Etheridge, 2012). The ALR is often scaled by the volume of the building or an area, such as the heated floor area in Denmark (Etheridge, 2012), where it is known as the Specific Air Leakage (BSI 2000), or in Finland and the UK where the ALR is scaled by the envelope area and is known as the Air Permeability (CIBSE, 2003; BSI, 2000). Because operational pressure differences are dynamic and normally an order of magnitude lower, at around 4Pa (Etheridge, 2012), the metric of ALR is only a physical property of a building that indicates the resistance of its fabric. Thus there is much uncertainty when using a value of ALR in an attempt to predict infiltration. The ALR at a pressure differential of 50Pa, $\dot{V}_{50}$ (m^3/s), must be converted to an operational infiltration rate, $\dot{V}_I$ (m^3/s), and although there are several approaches for converting $\dot{V}_{50}$ to $\dot{V}_I$ (Younes *et al.*, 2012) the most common *rule-of-thumb* for dwellings is given by Sherman (1987) as:

$$\dot{V}_{50} / \dot{V}_I = N \quad (1)$$

Equation (1) is often known as the *Sherman's ratio* or the *leakage-infiltration ratio* (Sherman, 1987), but N is frequently taken to have a value of 20 when the relationship is referred to as the *rule-of-20*. However, the figure of N=20 must not be viewed as fixed and should be scaled according to a variety of factors such as dwelling height, shielding, air leakage path size, and climate (Sherman, 1987). In the UK, the Standard Assessment Procedure (SAP) is the government's method for assessing and comparing the energy and environmental performance of dwellings used to make energy and environmental policy decisions. As a starting point, SAP applies Equation (1) to obtain an initial rate of infiltration from measured ALR. It then adds extra infiltration if chimneys, flues, and fans are present in a dwelling. This revised figure of infiltration is scaled if local shielding or mechanical ventilation are present. Other building codes make similar assumptions (MoEoF, 2012).

Unfortunately, these building codes do not scale an estimation of infiltration to take account of the permeability of *party walls*, here defined as walls shared by conjoined buildings. In order to predict heat lost via exfiltration from a conjoined dwelling under operational conditions, one must differentiate between the infiltration of unconditioned ambient air that requires heating, and conditioned air from an adjacent dwelling that does not. Accordingly, it is important to determine the permeability of party walls when measuring air leakage at a pressure differential of 50Pa so that infiltration under operational conditions through external façades and the energy required to heat it can be predicted.

The relevant literature reveals that little attention is paid to the measurement of airflow through party walls during air leakage tests, although techniques and protocols for its assessment exist (Feustel, 1990; Fernández-Agüera *et al.*, 2011). Measurements of air leakage through party walls separating a series of terraced houses and apartments have indicated that such flows can be a significant component of total air leakage rate at a pressure differential of 50Pa – up to 30% (Stephen, 1998). In Minnesota, USA, party wall airflow in apartment blocks is shown to contribute 1-65% of the total air leakage rate at a pressure differential of 50Pa (Bohac *et al.*, 2007), and in Korean high rise apartments it comprise 42-68% of the total air leakage rate at a pressure differential of 50Pa (Yun *et al.*, 2012).

In the UK, for example, ~80% of the housing stock shares at least one wall with another dwelling (DCLG, 2011). Thus this paper addresses the issue via a modelling based approach. In Section 2 extreme assumptions about the permeability of party walls of a conceptual apartment are discussed, and a simple but useful model of adventitious infiltration for a conjoined dwelling is presented. Then, the predictions of the infiltration model are corroborated against those of CONTAM and AIDA, two independent validated airflow analysis tools, for an identical building and environmental conditions. In Section 3, the infiltration model is used to predict the mean infiltration rate and the corresponding energy required to replace heat lost via operational exfiltration during the heating season for an apartment and a terraced house located in 14 different UK cities. Finally, in Section 4 the predictions of the infiltration model are used to develop a relationship between the adventitious air leakage rate under pressure, and operational infiltration and energy consumption during the heating season.

2. Method

In this paper, airflow through permeable party walls (façades 3-6 in Figure 1) is not considered under operational conditions because adjacent dwellings are assumed to experience identical environmental conditions and thus have the same internal pressure; this is discussed in Appendix 1. Therefore, airflow is only considered through external façades (façades 1-2 in Figure 1) and not internal walls. Conversely, when undertaking a standard measurement of air leakage in a conjoined dwelling of interest, it is realistic to consider that adjacent dwellings will not be undertaking a similar test simultaneously and so two extreme assumptions about the permeability of party walls can be made: A(1) party walls are permeable and so airflow to adjacent dwellings through them will occur (airflow through façades 1-6 in Figure 1); or A(2), party walls are impermeable

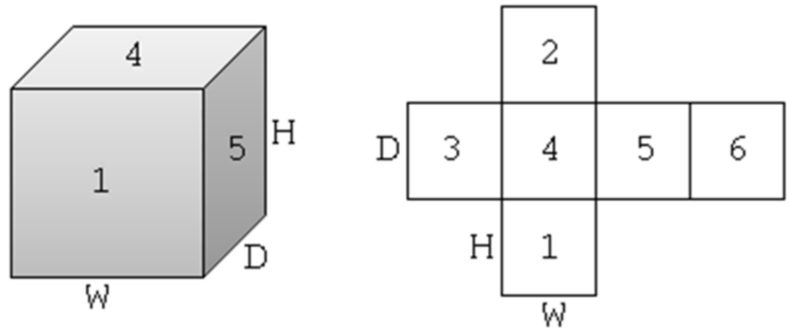

Figure 1. Archetypal conjoined dwelling of height H, width W, and depth D. Façades 1 and 2 are exposed and 3-6 are internal party walls. Airflow through party walls 3-6 is not considered under operational conditions but may be considered when making a measurement of air leakage according to two assumptions: A(1): façades 1-6 are permeable; A(2): only façades 1-2 are permeable.

and so no airflow to adjacent dwellings will occur (airflow through façades 1-2 in Figure 1). Accordingly, this paper asks the questions: what are the consequences of these two extreme assumptions of permeability and how do they affect Equation (1)? To help answer these questions we apply a two-dimensional model of adventitious infiltration for a conjoined dwelling.

2.1 DOMVENT: A 2D Integrating Infiltration Model

In the absence of knowledge of the location of air leakage paths (ALPs), we start by assuming that a wall is uniformly porous (Persily *et al.*, 2010). The modelling of wind driven infiltration using an envelope flow model, such as CONTAM or AIDA (Walton and Dols, 2005; Orme and Leksmono, 2002), is simple because a single flow path, representative of all ALPs, is placed at an arbitrary height on each façade. Modelling buoyancy is more problematic, but guidance on the number and location of ALPs is given by the AIVC (Orme and Leksmono, 2002) which states that "the simplest approach would be to assign a high positioned and low positioned leakage path to each façade." However, they also note that "we have found that 11 vertical holes, equally spaced, are required to model the stack flow though a uniformly porous wall to an accuracy of 3-4%", although no evidence is given showing why 11 ALPs is an optimum number. The greatest error occurs when buoyancy forces are introduced into an infiltration model and so we propose a framework in which the pressure difference across each section of the thermal envelope of a dwelling are estimated explicitly and the resulting airflow rates integrated over the whole envelope to give a total infiltration rate. This approach offers a coherent starting point to investigate infiltration in naturally ventilated dwellings and so is utilized here. To the best of our knowledge, this type of model has not been used to investigate infiltration in naturally ventilated dwellings and so its application is considered novel. We directly apply the work of Lowe (2000) and Lyberg (1997) whose 2D Integrating Infiltration Model is herein known as DOMVENT. Full details of the model are given by Lowe (2000). The simplicity of the DOMVENT model means that calculation time is significantly less than that for conventional airflow analysis tools, such as CONTAM and AIDA, which use a large number of defined ALPs. DOMVENT is thus a useful tool for undertaking the simulations necessary to investigate the infiltration one might expect to find in a conjoined dwelling subjected to varying climatic conditions.

A dwelling can be treated as a single-zone space by assuming that its rooms are interconnected and all internal doors are open (Etheridge, 2012). Then, mass conservation ensures that the net mass flow rate $\dot{M}_{net}$ (kg/s) of air through the thermal envelope of a dwelling of height H (m) is zero, and is given by:

$$\dot{M}_{net} = \int WEF\left(\left|\Delta p\right|\right)\varepsilon\left(\Delta p\right)dz + \dot{M}_{fan} = 0 \qquad (2)$$

where W is the dwelling width, E is the dimensionless relative leakage area, F is a flow function (kgm^{-2}s^{-1}), Δp (Pa) is the pressure difference across an infinitesimal section dz (m) of the thermal envelope in the vertical plane, the flow

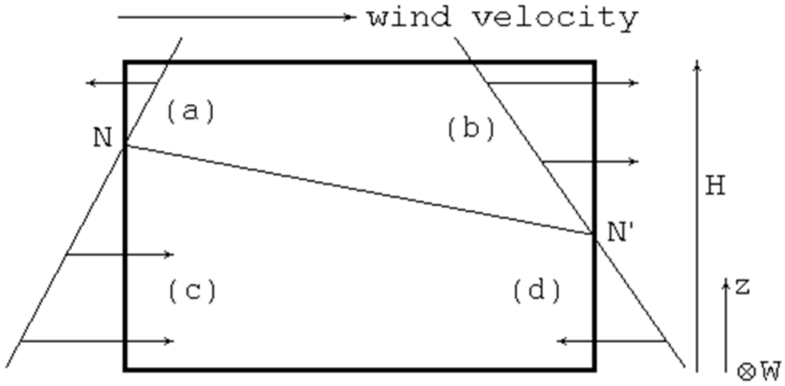

Figure 2. Vertical cross section through a dwelling showing stack pressure gradients on the windward and leeward façades and airflow modes: (a) windward exfiltration; (b) leeward exfiltration; (c) windward infiltration; (d) leeward infiltration. Line NN' is the neutral plane within the dwelling whose vertical deviation is caused by the action of wind around the dwelling. W is the dwelling width extending into the page.

direction function $\varepsilon(x) = 1$ if x>0, -1 if x<0, or $\varepsilon(x) = 0$ if x=0, and $\dot{M}_{fan}$ is the mass flow rate (kg/s) supplied by a mechanical fan. The model assumes that there is no airflow through horizontal and vertical party walls under operational conditions and so infiltration only occurs through two opposite façades, and that each façade is uniformly porous. Figure 2 shows the stack pressure gradients on the windward and leeward façades and the neutral points (N and N') where there is no airflow through the envelope. The heights of N and N' (above ground) are affected by the action of the wind around the dwelling. Air flows into the dwelling below the neutral points and out above them, giving up to four airflow modes: (a) windward exfiltration; (b) leeward exfiltration; (c) windward infiltration; (d) leeward infiltration. When integrating Equation (2), the limits of integration are set to account for each of these four modes between z=0 m and z=H m, see Lowe (2000).

The flow function of Equation (2) has the form:

$$F = m\left(\left|\Delta p\right|\right)^{n} \quad (3)$$

where n is the flow exponent with a value in the range of 0.6-0.7 (Orme and Leksmono, 2002), although it is often taken as 0.5 to simplify the analysis when $m=(2\rho)^{0.5}$ where ρ is the air density (kg/m^3). Otherwise, m corresponds to $\rho(2/\rho)^n$. We note that the power law relationship is considered to be less accurate than the quadratic relationship at operational pressure differences (Etheridge, 2012 Section 3.5.1), but it is the most widely used method of interpolating between measurements of ALRs (CIBSE, 2000; Sherman, 1987) and so it is employed here. The permeability of a building is normally recorded at a pressure differential of 50Pa and under these conditions Lowe (2000) shows that Equation (2) becomes:

$$\dot{M}_{50} = EmA_{50}(50)^{n} = \dot{M}_{fan}. \quad (4)$$

Here, $\dot{M}_{fan}$ is air supplied (kg/s) by a blower-door fan to achieve a pressure differential of 50Pa and A_{50} (m^2) is the area of the envelope able to transfer mass at a pressure differential of 50Pa. When permeability assumption A(1) is applied $A_{50} = A_{env}$, the area of the dwelling envelope (façades 1-6 in Figure 1). When permeability assumption A(2) is applied $A_{50} = A_{exp}$, the total area of the exposed façades (façades 1-2 in Figure 1). Equation (4) is used to calculate E for the whole dwelling.

2.2 Validating DOMVENT

DOMVENT is used to answer the questions posed by this paper (what are the consequences of the two extreme assumptions of permeability and how do they affect Equation (1)?) by predicting infiltration through the thermal envelope of a number of dwellings. Therefore, it is important to have confidence in the predictions of DOMVENT and so they are compared against those of established envelope flow models. The first is CONTAM, a validated multi-zone ventilation and pollutant transport model (Walton and Dols, 2005), and the second is the AIDA algorithm, a simple single-zone

Table 1. Properties of an archetypal apartment (Shurbsole et al. 2012) and terraced house (Oikonomou et al. 2012).

Dwelling Parameter	Apartment	Terraced House
Width, W, height, H, depth, D (m)	7.8, 2.6, 7.0	6.2, 5.6, 10.5
Envelope area, A_{env} (m^2)	186.16	317.24
Total exposed façade area, A_{exp} (m^2)	40.56	69.44
Party wall area, A_{env}–A_{exp} (m^2)	145.60	247.80
Air permeability (m^3/h/m^2)	10	10
ACH_{50} (h^{-1})	13.11	8.68
Relative leakage area, $E_{A(1)}$, $E_{A(2)}$	1.43×10^{-4},6.55×10^{-4}	1.43×10^{-4}, 6.52×10^{-4}
Wind scaling height (m)	5.45	5.60

ventilation model (Orme and Leksmono, 2002). CONTAM and AIDA are chosen to facilitate multi-zone (see Appendix 1) and automated investigations of infiltration, respectively. Both models assume a power law relationship between volume flow rate of air through the i^{th} ALP of a total of j ALPs and the pressure difference across it

$$\dot{V}_i = C_i \left(\Delta p\right)^n \tag{5}$$

where C_i (m^{3}s^{-1}Pa^{-n}) is a flow coefficient for the i^{th} opening. Full descriptions of the models are given in their respective references. All of the models discussed here assume that energy and mass conservation is observed, flow characteristics are constant in the mean, the zone is perfectly mixed, and internal air velocities are negligible and do not affect the internal hydrostatic pressure (Etheridge, 2012).

To help compare the predictions of the models the dimensions of a naturally ventilated archetypal UK apartment are used (Shrubsole *et al.*, 2012), see Table 1. The apartment has a floor area and height of 54.6 m^2 and 2.6 m, respectively, two exposed façades (see façades 1-2 in Figure 1) oriented north-south each with an area of 20.3 m^2, an envelope area of 186.2 m^2, and an air permeability of 10 m^3/h/m^2, the maximum permissible for a new UK dwelling (HM Government, 2010). Accordingly, using permeability assumption A(1), E=1.43×10^{-4} (calculated using Equation (4) and given in Table 1) and the ALR through each exposed façade at a pressure differential of 50Pa is the product of the air permeability and the façade surface area. Therefore, a standard flow coefficient for each exposed façade, C_f, is calculated using Equation (5), when j=1, to be C_f=10×20.3/(50$^{0.66}$×3600)=0.0043 m^{3}s^{-1}Pa^{-n}, where the flow exponent n=0.66, a typical value for ALPs (Orme and Leksmono, 2002). Windward and leeward façade pressure coefficients are 0.603 and −0.452, respectively, and are specifically for a long wall (Swami and Chandra, 1987). Predictions are made assuming an air density of 1.21 kg/m^3 for two conditions: wind only, and buoyancy only. These conditions require that no mechanical ventilation is present, so $\dot{M}_{fan} = 0$.

To model the wind only scenario using CONTAM and AIDA, a single ALP is located at the centre of each façade and wind speed is varied from u=1 m/s to u=5 m/s. When compared to DOMVENT for all wind speeds, the predictions of CONTAM are 0.23% lower at all wind speeds, whereas the predictions of AIDA are 0.04% higher. These models predict wind pressure in the same way and so one would not expect to see big differences

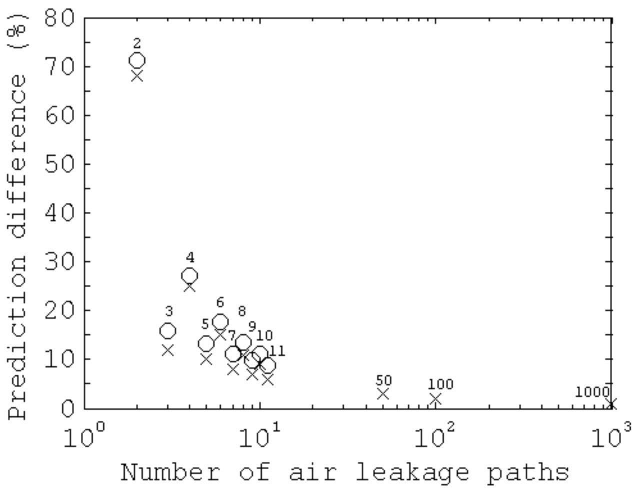

Figure 3. Percentage difference between the predictions of CONTAM and AIDA and DOMVENT for a varying number of ALPs. ×, AIDA; ○, CONTAM; Number of ALPs. Wind velocity, 0 m/s; ΔT=10°C; H=2.6 m.

between their predictions. Variation may be attributed to the different numerical solving techniques and rounding errors.

The buoyancy only scenario is also modelled using CONTAM and AIDA where 2 to 11 equally spaced ALPs are placed at heights z=0 m to z=H m at intervals of $H(j-1)^{-1}$ metres (where j is the number of ALPs). The internal temperature T_{int} (°C) is arbitrarily set at 19°C and the flow coefficient for each path is given by $C_i = C_f/j$. When compared to DOMVENT, for ΔT=10°C, the predictions made with 2 paths using CONTAM are 71% higher, whereas the predictions of AIDA are 68% higher. This overestimation by the modelling tools in relation to DOMVENT is expected because the distance between the paths is the maximum possible and is equal to H. Therefore, increasing the number of paths systematically to 11, and reducing their separation, decreases the difference between the predictions of CONTAM and AIDA and those of DOMVENT, see Figure 3. When compared to DOMVENT for a temperature difference of 10°C, the predictions made with 11 paths using CONTAM are 8.8% higher, whereas the predictions of AIDA are 5.9% higher. This inter-model comparison demonstrates reassuring agreement between their predictions.

Based on the increased confidence in the predictions of DOMVENT and, as an interesting aside, we ask the question: what is an optimum number of ALPs when modelling infiltration using an envelope flow model that balances prediction accuracy with calculation time? For this study, the data input of ALPs into CONTAM is done manually through the CONTAMW interface whereas data input into AIDA is automated using bespoke MATLAB code (MathWorks, 2011). Using AIDA the number of ALPs on each façade is increased successively to 50, 100, and 1000, and its predictions are reduced to 2.6%, 1.8% and 1.22%, respectively, above those of DOMVENT. This analysis suggests the difference between the predictions of the models reduce as the number of paths located on each façade approaches infinity asymptotically, but with diminishing returns, see Figure 3. However, for all practical purposes, 11 paths is close enough to infinity for a reasonably accurate prediction of buoyancy driven infiltration (<6% difference) using a conventional envelope flow model such as CONTAM or AIDA, thus substantiating AIVC guidance on infiltration modelling (Orme and Leksmono 2002).

Figure 3 shows that an odd number of ALPs gives better agreement than an immediately higher even number. An odd spacing places an ALP at the neutral height where the pressure difference across it and airflow through it is zero. Thus, the porosity of the wall reduces and better agreement is achieved, albeit artificially. Increasing the number of paths reduces the effect of this anomaly.

3. Results

The CIBSE Test Reference Year (TRY) weather data set (CIBSE, 2002) is a synthesised typical weather year suitable for analysing the environmental performance of buildings in the UK. Data exists for 14 locations, coastal and inland, varying in latitude from 50.35°N to 55.95°N and longitude from 6.22°W to 1.36°E. Accordingly, these data are applied to the archetypal apartment (now considered to be located on the 1st floor) between 1st October and 1st March, thus simulating the heating season when purpose-provided ventilation is at a minimum. The rate of heat loss (W) via infiltration is given by:

$$H(t) = \dot{M}_{inf} c \Delta T \quad (6)$$

where c is the specific heat capacity of air (c=1004Jkg^{-1}K^{-1}) and ΔT (K) is the difference between the internal and external air temperatures. ΔT is evaluated with internal air temperature T_{int}=18.5°C when external air temperature T_{ext}≤15.5°C. Otherwise, following Lowe (2000)

$$T_{int} = T_{ext} + 3e^{-(T_{ext}-T_{base})/3} \quad (7)$$

to account for the tendency of T_{int} to increase at the beginning and end of the heating season as T_{ext} rises. Equation (7) employs a *base temperature* (CIBSE, 2006) of T_{base}=15.5°C; this is chosen because the heating system of an average UK dwelling begins to operate when T_{ext}≤(T_{int}−3°C) (Hamilton *et al.*, 2011). Accordingly, the rate of heat loss is not recorded when T_{ext}>15.5°C because it is assumed that the heating system is off. Heat loss (kW) is estimated over periods of t=1 hour and so it is easily converted to the total energy lost by operational infiltration (kWh).

Table 2. Predicted infiltration air changes per hour (h^{-1}) and total heat loss (kWh) during heating season in an archetypal apartment for two limiting permeability assumptions. Air permeability, 10m^3/h/m^2; A_{env}:A_{exp}, 4.59.

Location	Assumption A(1): Permeable party walls				Assumption A(2): Impermeable party walls			
	mean ac/h	median ac/h	σ ac/h	total heat loss	mean ac/h	median ac/h	σ ac/h	total heat loss
Belfast	0.16	0.13	0.12	464	0.75	0.61	0.53	2131
Birmingham	0.14	0.10	0.10	379	0.64	0.46	0.47	1740
Cardiff	0.15	0.12	0.11	395	0.69	0.56	0.50	1811
Edinburgh	0.15	0.10	0.12	415	0.67	0.45	0.56	1906
Glasgow	0.15	0.10	0.12	436	0.68	0.44	0.57	2002
Leeds	0.10	0.07	0.07	278	0.47	0.32	0.34	1278
London	0.12	0.09	0.09	299	0.57	0.40	0.43	1370
Manchester	0.14	0.10	0.11	389	0.66	0.48	0.51	1785
Newcastle	0.11	0.08	0.09	321	0.52	0.37	0.40	1475
Norwich	0.15	0.11	0.12	409	0.70	0.50	0.56	1875
Nottingham	0.13	0.10	0.09	387	0.61	0.47	0.42	1774
Plymouth	0.20	0.15	0.17	430	0.91	0.70	0.77	1975
Southampton	0.08	0.06	0.06	211	0.37	0.28	0.25	968
Swindon	0.17	0.13	0.13	465	0.76	0.58	0.59	2135
TOTAL	**0.14**	0.10	0.11	mean 352	**0.64**	0.45	0.52	mean 1615
$\dot{V}_{50}:\dot{V}_I$	**93.6**				**20.4**			

Wind speed is scaled for an urban environment using the power law formula with a coefficient of 0.35 and an exponent of 0.25 (BSI, 1991). Façade wind pressure coefficients are varied according to the wind direction using the distribution described in Section 2.2 (Swami and Chandra, 1987). The relative leakage area is varied according to the two permeability assumptions so that under A(1), $A_{50}=A_{env}$; and under A(2) $A_{50}=A_{exp}$. Therefore, $E_{A(1)}=1.43\times10^{-4}$ and $E_{A(2)}=6.55\times10^{-4}$, respectively, see Table 1. Air density is 1.23kg/m^3, $\dot{M}_{fan}=0$, and all other variables are given in Table 1.

Table 2 gives the predicted mean, median, and standard deviation (σ) infiltration rate in air changes per hour (ac/h) in an archetypal apartment during the heating season for the two extreme permeability assumptions in each city and overall. Also given is total heat loss (kWh) for each city and the overall mean value. For this example, if permeable party walls are assumed, the infiltration rate is below 0.5 ac/h, which is recommended by many European countries as a threshold ventilation rate above which some negative health effects reduce (Dimitroulopoulou, 2012). Under these circumstances, additional purpose-provided ventilation would be required. If impermeable party walls are assumed, the opposite is true, highlighting the importance of the assumption about the behaviour of party walls. Table 2 also shows, when party walls are considered to be permeable, the ratio of airflow rates at pressure to those under operational conditions is 93.6, which is much greater than that given in Equation (1), whereas when party walls are considered to be impermeable the ratio is 20.4, which is very close to that given in Equation (1). This suggests that Equation (1) was originally formulated from measurements made in dwellings that either had no party walls or impermeable party walls, and required little scaling. A rough sensitivity analysis of the model shows that rotating the apartment through 90° increases average infiltration rates by 7% and so the simulations obtained here stand.

DOMVENT is now used to assess the infiltration rate of a UK terraced house (Oikonomou *et al.*, 2012) using the properties given in Table 1 and CIBSE TRY weather data. The assumptions of airflow through party walls under operational conditions and when making a measurement of air leakage are identical to those for the apartment (see Figure 1). In a similar pattern to that of the apartment, the mean infiltration rate during the heating season is predicted to be 0.1 h^{-1} when party walls are considered permeable (see façades 3-6 in Figure 1), and 0.47 h^{-1} when they are not, see Table 3. The leakage-infiltration ratios are predicted to be 84.9 and 18.6, respectively. Rotating the terrace through 90° increases the infiltration rate by 6%. Accordingly, these predictions for an archetypal terrace house confirm the patterns of infiltration behaviour identified by the analysis of an archetypal apartment.

4. Discussion

Consideration of Equation (4) demonstrates that the predictions made by DOMVENT for the two assumptions of party wall permeability are related by a simple ratio of the two effective leakage areas $E_{A(2)}$:$E_{A(1)}$, and by the ratio of exposed façade area to envelope area A_{env}:A_{exp}; they both give the same value. Accordingly, the predictions made assuming permeable party walls can easily be scaled to identify those for impermeable part walls. For example, converting from the predicted mean infiltration ac/h for the apartment for permeable party walls (see Table 2) to that for impermeable party walls is $ac/h_{A(2)}=ac/h_{A(1)}(A_{env}:A_{exp})=0.14(186.16/40.56)=0.64h^{-1}$; see Table 1 for values of A_{env} and A_{exp}. Furthermore, if the exposed façades and party walls are not equally porous but the proportion of air leakage through each surface is known, it is straightforward to amend a prediction of mean infiltration ac/h made for assumption A(2). For example, Stephen (1998) suggests that up to 70% of air leakage occurs through exposed façades in UK apartments and terraced houses. Then, Equation (4) shows that E is 70% of the value calculated for A(2); see Table 1. For the archetypal apartment, the amended mean ac/h is $0.7\times0.64=0.45h^{-1}$ (see Table 2) and the leakage-infiltration ratio is 13.1/0.45=29.13. This means that Equation (1) can be amended according to one's knowledge of party wall permeability. The ability to scale infiltration rate also means that it is also possible to scale predictions of total energy loss.

In Figures 4 and 5 distributions of predicted infiltration rate are shown in the archetypal apartments and house, respectively. Distributions are given for each of the 14 locations of the CIBSE TRY weather data set and an overall distribution encapsulates predictions of infiltration in all locations (ALL). The upper bar denotes the maximum, the lower bar the minimum, and the central bar the median. The box denotes the inter-quartile range, and the cross denotes the mean

Table 3. Predicted infiltration air changes per hour (h^{-1}) and total heat loss (kWh) during heating season in an archetypal terraced house for two limiting permeability assumptions. Air permeability, $10m^3/h/m^2$; A_{env}:A_{exp}, 4.57.

Location	Assumption A(1): Permeable party walls				Assumption A(2): Impermeable party walls			
	mean ac/h	median ac/h	σ ac/h	total heat loss	mean ac/h	median ac/h	σ ac/h	total heat loss
Belfast	0.12	0.09	0.07	867	0.53	0.40	0.33	3963
Birmingham	0.10	0.08	0.06	735	0.47	0.34	0.29	3360
Cardiff	0.11	0.08	0.07	733	0.49	0.37	0.31	3349
Edinburgh	0.11	0.08	0.08	811	0.49	0.34	0.34	3705
Glasgow	0.11	0.08	0.08	855	0.50	0.35	0.35	3907
Leeds	0.08	0.07	0.04	569	0.36	0.30	0.20	2601
London	0.09	0.07	0.06	584	0.42	0.32	0.26	2669
Manchester	0.10	0.08	0.07	741	0.48	0.34	0.32	3384
Newcastle	0.09	0.07	0.05	634	0.39	0.31	0.24	2897
Norwich	0.11	0.08	0.08	772	0.50	0.35	0.35	3529
Nottingham	0.10	0.08	0.06	742	0.44	0.34	0.25	3390
Plymouth	0.14	0.10	0.11	790	0.64	0.46	0.49	3610
Southampton	0.07	0.06	0.03	460	0.30	0.27	0.15	2103
Swindon	0.12	0.08	0.08	863	0.54	0.38	0.37	3944
TOTAL	**0.10**	0.07	0.07	mean 677	**0.47**	0.34	0 .32	mean 3094
$\dot{V}_{50}:\dot{V}_I$	**84.9**				**18.6**			

average. It is interesting to note that the distribution does not change according to one's assumption of party wall permeability; the left hand y-axis shows predicted infiltration for permeability assumption A(1), permeable party walls, and the right hand y-axis for permeability assumption A(2), impermeable party walls. Figures 4 and 5 show that the distributions are right skewed and that infiltration rates in the apartment and house are more likely to be closer to the median than the mean.

An analysis of the heights of the neutral points shows that the windward exfiltration and leeward infiltration modes (see Figure 2) are not present for 82% and 74% of the time in the apartment and house, respectively. Accordingly, infiltration in both dwelling types is predicted to be dominated by wind driven forces via windward infiltration and leeward exfiltration modes. This is more likely in the apartment because its external façade height is less than that of the house, and so the difference between the heights of its windward and leeward neutral points only has to be ≥2.6m to eliminate the windward exfiltration and leeward infiltration modes. Furthermore, Figures 4 and 5 show that the shape of each distribution is influenced by the local scaled wind speed; for example, Plymouth (Ply) has the greatest mean scaled wind speed of 3.2m/s and the greatest standard deviation of 1.9m/s, whereas Southampton (Sou) has the lowest mean scaled wind speed of 1.5m/s and the lowest standard deviation of 1.0m/s. Similarly, Plymouth has the largest inter-quartile range and Southampton the smallest.

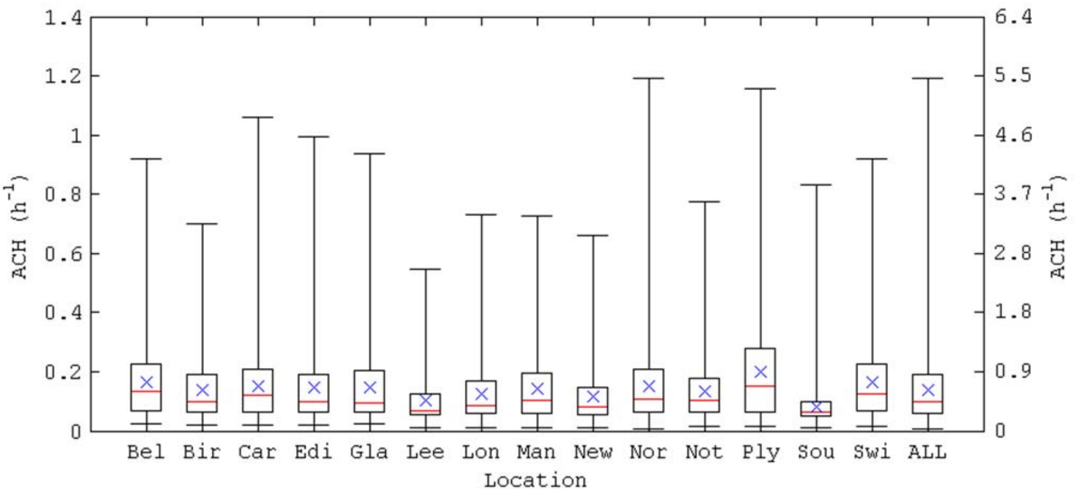

Figure 4. Predicted infiltration air change rate (h^{-1}) during heating season in an archetypal apartment with permeable party walls (left hand y-axis) and impermeable party walls (right hand y-axis). Air permeability, $10m^3/h/m^2$. ×, location mean.

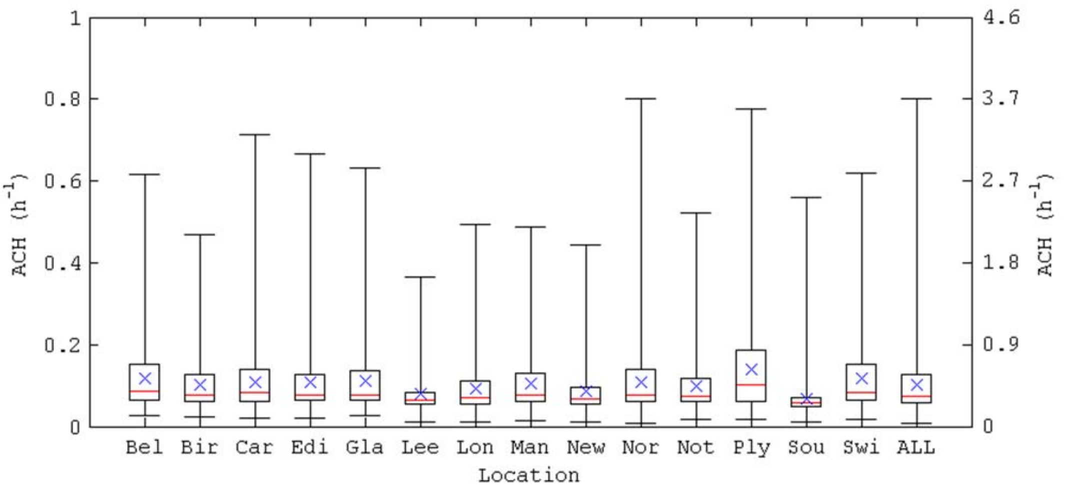

Figure 5. Predicted infiltration air change rate (h^{-1}) during heating season in an archetypal terraced house with permeable party walls (left hand y-axis) and impermeable party walls (right hand y-axis). Air permeability, $10m^3/h/m^2$. ×, location mean.

There are several consequences of these findings. If permeability assumption A(1) is true and party walls are indeed permeable, then conjoined dwellings do not experience the rate of operational infiltration predicted by Equation (1). Accordingly, annual energy lost via operational infiltration is also less than one might expect, see Tables 2 and 3. This is important for current policies aimed at increasing the energy efficiency of the housing stock (required to mitigate greenhouse gas emissions) since the payback period of retrofitted energy efficient measures designed to increase the air tightness of a conjoined dwelling would increase dramatically. The lower than expected infiltration rates could also have health consequences by allowing the build-up of pollutants from internal sources, such as fine particulate matter (Shrubsole *et al.*, 2012), moisture, carbon monoxide, and radon (Pacheco-Torgal, 2012). However, if permeability assumption A(2) is true and party walls are already impermeable then a sensible energy efficiency measure is the tightening of exposed façades. Although the dwelling types and weather data applied here are from the UK, the findings can be applied by the policy makers of any country with a large number of conjoined dwellings and for those building codes that apply Equation (1) in some form.

5. Conclusions

This paper presents an analysis of infiltration rates in conjoined dwellings based on two extreme assumptions of party wall permeability at high pressure. The first assumption assumes that party walls are permeable, and in this instance the leakage-infiltration ratio is predicted to be significantly greater than that used by building codes to evaluate the energy and environmental performance of dwellings. The second assumption assumes that party walls are impermeable and here the leakage-infiltration ratio is predicted to be close to that used in practice. With this knowledge, it is now possible to amend the leakage-infiltration ratio for a given application, and to use it to make informed decisions on the implementation of energy efficiency measures. These findings have significant energy and health implications and should be of great interest to the policy makers of any country with a large number of conjoined dwellings. Finally, the paper also provides evidence for AIVC guidance on the modelling of infiltration using envelope flow models where none existed previously.

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Appendix 1: Infiltration in Tall Apartment Blocks

The English Housing Survey (EHS) is a statistically representative survey of over 16,000 English dwellings (DCLG, 2011). The tallest surveyed apartment block has 39 storeys and a height of 103.1m, but the EHS also estimates the mean number of storeys in an apartment block to be 3.6 with a mean block height of 9.3m. While English apartment blocks are quite modest in height, those with a height greater than 100m are common in Asian cities (Yun *et al.*, 2012). However, it is reasonable to expect that a dwelling can be located on any level within a block of apartments and this should be considered when modelling infiltration.

In order to simplify the DOMVENT model, some assumptions are made about a dwelling and its location within a block. A conjoined dwelling, such as an apartment, is assumed to be joined to four immediately adjacent apartments and a semi-infinite number of other apartments in both the vertical and horizontal planes. In the horizontal plane each dwelling is a mirror image of its adjacent apartment, whereas in the vertical plane each dwelling is identical to that located above and below it. Under operational conditions, with all purpose-provided openings sealed, the DOMVENT model does not consider airflow between adjacent dwellings through permeable party walls because it assumes that they experience identical environmental conditions and thus have the same internal pressure.

While this assumption is valid in the horizontal plane, it may be invalid in the vertical plane where hydrostatic pressure varies with height. In order to investigate the effect of a change in the vertical location of an archetypal apartment (for dimensions see Table 1) within a block on the overall infiltration rate of unconditioned air, CONTAM is used to create two models. The first CONTAM model is an airflow network of 50 apartments stacked one on top of the other with a 0.25m inter-floor separation. ALPs are located at floor and ceiling height in the two vertical external façades and in the two horizontal party walls (floor and ceiling). The second CONTAM model comprises a single zone with ALPs located at floor and ceiling height in the two vertical external façades only.

Figure 6 shows the difference between the predicted rate of infiltration of unconditioned air into apartments with permeable horizontal party walls located within a 50 storey block and a single zone apartment with impermeable horizontal party walls. Each apartment is subjected to environmental conditions of ΔT=10°C and u=0m/s, and has an air permeability of 10m^3/h/m^2. The predicted infiltration rate in the apartments with ALPs located in horizontal party walls generally increases with height, but Figure 6 also shows anomalies at each end of the block. The flow coefficients for each

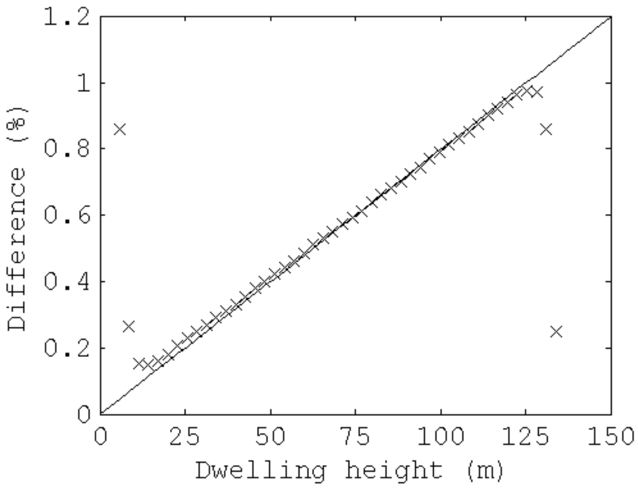

Figure 6. Difference between predicted infiltration rates of unconditioned air into apartments with permeable horizontal party walls located within a 50 storey block and a single zone apartment with impermeable horizontal party walls. ΔT=10°C and u=0m/s. Linear fit through origin to floors located between 14.25m and 125.15m; Gradient=8% per km, R^2=0.99.

ALP are estimated using Equation (5) and the method described in Section 2.2, but this approach increases the permeabilities of apartments located towards the top and bottom of the block, relative to those located in the middle, because they have less adjacent apartments that offer resistance to inter-dwelling airflow. Here, we assume that it is reasonable to expect there to be no difference between the predictions of the two models (with and without permeable horizontal party walls) at ground level and so we ignore the outliers and fit a straight line through the origin to floors 6 to 44 located between heights 14.25 m and 125.15 m, giving a gradient of 8% per km and a goodness of fit R^2=0.99. The estimations of difference presented here are independent of apartment permeability, and increase fractionally as the difference between the internal and external air temperatures decreases.

This analysis suggests that the vertical location of an apartment in a block does not significantly affect the overall infiltration rates of unconditioned air, particularly in the archetypal English apartment modelled here, and so when developing a model of infiltration one does not necessarily need to consider airflow through permeable horizontal party walls.

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Simulation of pollution transport in buildings: the importance of taking into account dynamic thermal effects

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Abstract
The recent introduction of the Generic Contaminant Model in EnergyPlus allows for the integrated modelling of multizone contaminant and dynamic thermal behaviour within a single simulation package. This article demonstrates how dynamic thermal simulation can modify pollutant transport within a building. $PM_{2.5}$ infiltration from the external to internal environment under dynamic thermal conditions is compared in CONTAM, EnergyPlus 8.0, and Polluto, an in-house pollutant transport model developed in EnergyPlus 3.1. The influence of internal temperature on indoor $PM_{2.5}$ levels is investigated by comparing results from standard CONTAM simulations and dynamic thermal EnergyPlus 8 simulations. Circumstances where the predictions of such models can diverge are identified.
Practical application: This technical note compares the performance of a new indoor air quality model in EnergyPlus, an EnergyPlus in-house model (Polluto), and an established model (CONTAM). The work then compares the results of indoor air quality models under static and dynamic internal temperature conditions, and demonstrates how predicted indoor pollution levels may deviate significantly if an inappropriate indoor temperature is used. Practically, the work provides confidence in the new models, as well as demonstrating the importance of having a good understanding of the thermal behaviour of a building when modelling indoor air quality.

Keywords
Indoor air, EnergyPlus, CONTAM, Polluto

Introduction

Airflow modelling is an essential tool in building design and analysis of indoor air quality. Air pollutants can be produced indoors by building occupants (e.g. water vapour, tobacco

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smoke, CO_2), the building envelope and internal furnishings (e.g. Volatile Organic Compounds), and microbial contaminants of the building (e.g. mould spores); these pollutants can circulate around a building and require removal from the internal air through appropriate ventilation. Additionally, pollutants from the external environment such as fine particulate matter ($PM_{2.5}$) may infiltrate the building and require removal. Existing airflow tools can include simplistic single-zone models, more complex multizone models, and highly complex computational fluid dynamics (CFDs) models. In multizone airflow models, buildings are treated as a series of nodes representing zones within the building, and connections between nodes representing airflow elements such as doors, cracks, and ducts. Two such models are CONTAM and the EnergyPlus Airflow Network, both of which now have the capability of modelling contaminant transport indoors.

CONTAM[1] is an air quality and ventilation analysis tool developed by the National Institute of Standards and Technology (NIST) that can be used to calculate airflow, contaminant transport through airflow, and building occupant exposure to contaminants. Airflow in CONTAM is, like the EnergyPlus Airflow Network, based on the AIRNET model,[2] while CONTAM has the additional capabilities of modelling contaminant concentrations inside buildings due to airflow-driven dispersal, adsorption and desorption to building materials, filtration, and deposition to building surfaces. CONTAM has been extensively validated[3–5] and offers a user-friendly means of understanding building ventilation and contaminant transport. CONTAM is strictly for airflow and contaminant analysis, and lacks the ability to simulate energy and thermal behaviour of buildings.

Under typical operating conditions, the thermal performance of buildings causes dynamic zonal air temperatures, which can, in some cases, have an important impact on the airflow through a building and therefore contaminant transport. Pressure losses across an airflow path can be described using an energy equation

$$\Delta P = (P_1 - P_2) - \left(\frac{\rho_1 v_1^2}{2} - \frac{\rho_2 v_2^2}{2}\right) + g(\rho_1 z_1 - \rho_2 z_2) \quad (1)$$

In equation (1), the first term represents static pressure differences, the second dynamic pressure differences due to wind, and the third differences due to buoyancy. The second term shows that at high wind speeds, the total pressure difference is dominated by the differences in static pressures and the wind speed. The last term shows that at low wind speeds the dominant effect is the density of the air on each side of the flow path. The density of air is typically calculated using dry air and water vapour components of the air, meaning that the air density depends on the temperature and the amount of moisture in the air, given by

$$\rho = \frac{P_d}{R_d T} + \frac{P_v}{R_v T} \quad (2)$$

Therefore, a change in the air temperature or the moisture content of air at low wind speeds can lead to a change in the pressure difference across a flow path, which in turn leads to a change in the movement of contaminants and water vapour.

While CONTAM is a very capable tool for calculating airflow rates and contaminant concentrations, a major limitation is that it is not a thermal modelling tool and so does not amend zonal air density in response to temperature changes due to building performance and occupant behaviour. There are a number of options available to CONTAM users who wish to account for the relationship between zonal air temperature and thermal performance of the building. The licensed dynamic thermal package TRNSYS has been coupled with CONTAM to enable combined airflow and heat transfer simulations for buildings, with bridging between the two tools allowing them to share simulation inputs and outputs.[6] Alternatively, CONTAM users can either define a time-variable internal air temperature estimated for the zone, import internal air temperatures from physically measured sources, or import temperatures from the outputs of a different dynamic thermal model, such as EnergyPlus.[7] Only the thermal modelling approaches can account for the complex interactions between the internal air temperature and building heat transfer mechanisms, such as ventilation convection, solar radiation, and fabric conduction. However, in uncoupled models this approach requires additional work to ensure that both models are identical, and adds significant time to the model development. In addition, the lack of feedback between two uncoupled models means that any changes to the CONTAM model that may impact internal temperatures (for example ventilation behaviour) require re-running thermal models and importing the new temperature schedules into CONTAM.

EnergyPlus[8] is an open-source whole building simulation program used for energy analysis and thermal load simulation. Airflow in EnergyPlus can be simulated using the multizone Airflow Network tool, an airflow model based on an early version of AIRNET and COMIS.[9] The model is capable of simulating infiltration and exfiltration into a building due to indoor/outdoor pressure differences, building envelope permeability, natural and mechanical ventilation, as well as zone-to-zone airflows. The Airflow Network capabilities of EnergyPlus have been validated by comparison with measured data,[10] and through an inter-model comparison with CONTAM.[11] The ability to model pollutant transport in EnergyPlus has recently been introduced in the form of the Generic Contaminant Model (GCM), which allows users to model a single contaminant within a building. This enables the modelling of coupled thermal behaviour and contaminant transport within a single simulation package. In addition, UCL has developed an in-house model, Polluto, which allows for contaminant transport modelling in EnergyPlus 3.1. EnergyPlus GCM currently has an advantage over Polluto in being able to use indoor contaminant concentrations as flags to alter building operation, for example allowing ventilation systems to operate above a certain concentration threshold. Conversely, Polluto is capable of modelling multiple pollutants, while the GCM is currently restricted to a single contaminant.

This article describes the simulation of a simple single-zoned building in CONTAM, the EnergyPlus 8.0 GCM, and EnergyPlus-Polluto. An intermodel comparison is made between the models for $PM_{2.5}$ infiltration into the building with dynamic internal temperatures derived from EnergyPlus. In addition, the results of simulations using dynamic internal temperatures are compared to simulations performed in CONTAM with constant internal temperatures, demonstrating scenarios where indoor air quality models decoupled to whole building energy models can perform unsatisfactorily.

Methodology

Model comparison

There are inherent differences between the calculation methods employed by CONTAM and EnergyPlus that need to be considered in comparing the model performances. The assumptions used by EnergyPlus and CONTAM to calculate local wind speed and therefore air pressures at the external entrances to airflow paths are different, although result, as one would expect, in similar values. Wind pressures provided in weather files are modified in both tools to account for the differences between the wind speeds at the weather station, and the expected wind speeds at the height and location of the building. Variables in both tools can be adjusted in order to give similar results.

In addition, EnergyPlus and CONTAM also both report their predictions differently: CONTAM gives instantaneous values for contaminant concentrations over the simulation period, while EnergyPlus gives integrated values. As a result, either smaller time steps need to be used with CONTAM or trapezoidal interpretation used on the results, particularly where there are rapid changes in contaminant concentration. In addition, EnergyPlus employs so-called 'WarmUp Days' which are run at the start of the EnergyPlus simulation to ensure that any thermal capacitance values are representative of the dwelling in the environment described by the weather file. Conversely, outputs from CONTAM are reported without an initial warm-up period, and so the concentrations of the two tools may not match over the initial period. For further information on the algorithms used by each program, see the CONTAM User Guide[1] and the EnergyPlus Engineering.[12]

Contaminant transport models are typically capable of modelling low-concentration trace contaminants (for example $PM_{2.5}$) and high-concentration non-trace contaminants (for example water vapour). $PM_{2.5}$ was chosen to be the contaminant of interest for this study. As it can be produced by occupant activities within a building as well as infiltrating into a building through controlled or uncontrolled ventilation, $PM_{2.5}$ has a strong airflow component when assessing occupant exposure.[13] Water vapour was also considered as a non-trace contaminant in order to include it in CONTAM air-density calculations.

To compare the performance of CONTAM with the EnergyPlus 8.0 GCM and EnergyPlus-Polluto, a single-zoned building ($4\,m \times 5\,m \times 2.8\,m$) with no windows, doors, heating, or pollution sources was created in the simulation tools. Airflow into the building was from infiltration through permeable walls ($3\,m^3\ h^{-1}\ m^{-2}$ @ 50 Pa), with the roof and floor considered impermeable. Building envelope materials were modelled to provide a U-value of approximately $0.5\,Wm^{-2}\,K$ by selecting appropriate materials. The wall permeability was simulated by installing crack-type flow paths near the bottom (0.05 m) and top (2.75 m) of each façade; the single-zoned building therefore has eight airflow paths. The cracks were modelled with a power law equation assuming one-way airflow.

For the chosen permeability of $3\,m^3\ h^{-1}\ m^{-2}$ @ 50 Pa, a flow coefficient (C) of $0.0004411\,m^3\ s^{-1}\,Pa^{-n}$ was used for the 5-m wall, and $0.0003529\,m^3\ s^{-1}\,Pa^{-n}$ for the 4-m wall. The flow exponent (n) for both walls was set to 0.66, as per Jones et al.[14] A Chartered Institution of Building Services (CIBSE) weather file for London Heathrow[15] was used for both simulation packages; simulations were run for winter (1 January to 21 January) and summer (1 July to 21 July) conditions. As CONTAM is unable to independently calculate dynamic internal temperatures of the building, a preliminary run was performed in EnergyPlus 8.0, and the predicted internal temperatures converted into a continuous value file (.cvf) to define the internal temperatures in CONTAM. Any moisture-buffering effects of the building envelope were ignored.

$PM_{2.5}$ was modelled with a molecular weight of $8\,kg\ kmol^{-1}$, with an initial internal concentration and constant external concentration set at $13\,\mu g\ m^{-3}$, a value approximately equal to the current mean outdoor $PM_{2.5}$ concentration in London.[16] $PM_{2.5}$ was removed from the internal air with a deposition rate of 0.39 (1/h) as per Ozkaynak et al.[17] Moisture was treated as a non-trace contaminant in CONTAM and was therefore included in air-density calculations. CONTAM airflow numerics were adjusted to account for the effect of flow element temperature on air density. The files were synchronised to ensure that they had exactly the same start times and time steps. Simulations were run with a reporting time step of 1 minute to minimise the discrepancy between the instantaneous output values of CONTAM and the integrated outputs of EnergyPlus. Zone air water content ratio (g/kg) and zone $PM_{2.5}$ concentration were output on a minute-by-minute basis over a 3-week period for the software tools, and the differences between them compared.

Limitations of uncoupled models

In order to accurately simulate contaminant transport in a building, room temperatures should be appropriate for the modelled building. To demonstrate the impact of using an inappropriate air temperature on the internal concentration of $PM_{2.5}$, the simulations were repeated in CONTAM with the internal temperature fixed. Internal temperatures were set to the average external temperature for the weather file during the three-week simulation period (2.7°C for the winter period, 18.7°C for the summer period). In addition, the simulations were run with the internal CONTAM temperature set to the average internal temperatures as predicted by EnergyPlus (4.3°C for the winter period, 25.9°C for the summer period) in order to account for internal gains caused by the thermal performance of the building envelope.

$PM_{2.5}$ and water content ratio were output on a minute-by-minute basis and the differences between the CONTAM model with static temperature and EnergyPlus GCM with a floating temperature analysed for the absolute differences. In addition, the relationship between external wind speed and internal $PM_{2.5}$ and RH differences between the two models was examined.

Results

The results showed a strong agreement with the EnergyPlus 8.0 GCM and the in-house EnergyPlus Polluto model, with an absolute deviation between the models of 0.09% for the winter ($\sigma = 0.09$) and 0.09% ($\sigma = 0.07$) for the summer simulation periods. Both the EnergyPlus GCM and Polluto model made $PM_{2.5}$ predictions similar to those of the CONTAM model when CONTAM was provided with dynamic EnergyPlus-derived temperatures (Figure 1). Differences between EnergyPlus GCM and CONTAM when run during the winter period averaged 5.0% ($\sigma = 3.7$); variations between the models may be attributable to different calculation methods and output reporting between the two tools. During the summer, differences between the model results for transient internal temperatures were found to decrease to 3.2% ($\sigma = 2.6$).

When internal temperatures were held constant in CONTAM, there were significantly larger differences between the predicted $PM_{2.5}$ concentrations between the two models (Figure 1). For winter simulations, the difference between the transient EnergyPlus and static internal temperatures are 9.9% ($\sigma = 10.2$) at 2.7°C and 7.1% ($\sigma = 8.4$) at 4.3°C – approximately double the difference under transient conditions. Under summer conditions, differences between the transient model and the static model are 12.8% ($\sigma = 15.1$) at 18.7°C and 7.0% ($\sigma = 7.0$) at 25.9°C. In both cases, the differences between the predictions are greater when the internal temperature was set at the average outdoor temperature rather than the average internal temperature as predicted by EnergyPlus; this emphasises the importance of accounting for the thermal performance of the building. When CONTAM simulations were run with the airflow numerics set to ignore the impact of temperature on air density, there were very large observed differences between transient and static CONTAM models and EnergyPlus at low wind speeds and summer high temperatures, indicating the importance of including air-density numerics under such scenarios.

Equation (1) describes how the influence of temperature may be most significant at low wind speeds and temperature-dependent buoyancy effects are dominant. The influence of wind speed on internal $PM_{2.5}$ concentrations for winter and summer simulations can be seen in Figure 2. At low wind speeds and higher temperatures (e.g. summer), the differences in predicted $PM_{2.5}$ levels between the models with static and transient internal temperature are highest.

Discussion

The results demonstrate a good agreement between CONTAM, the EnergyPlus GCM, and the in-house Polluto model when the model parameters are set to be the same in the programs, thus providing confidence in the EnergyPlus tools. Simulation results from

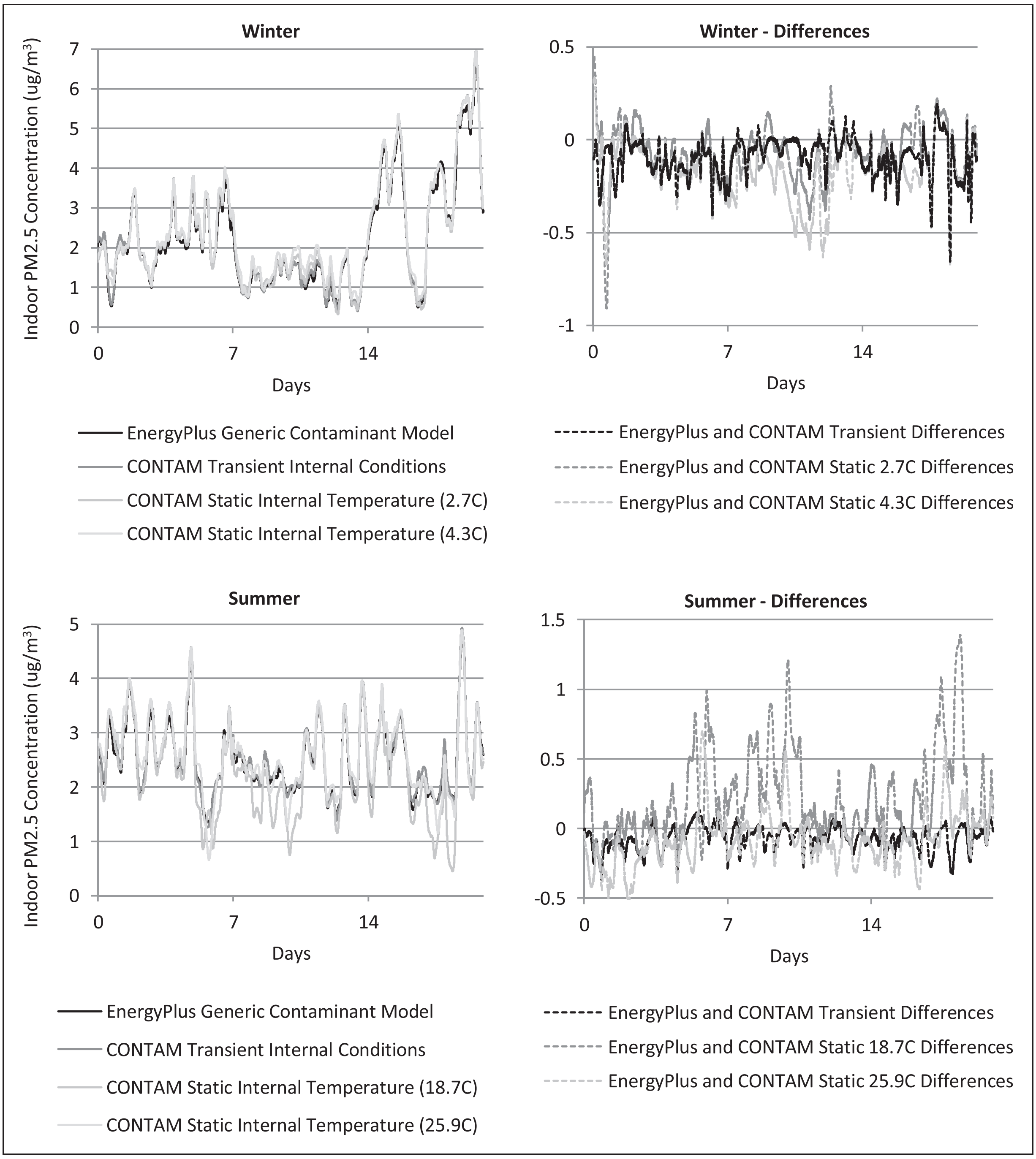

Figure 1. Indoor $PM_{2.5}$ concentrations for winter (top) and summer (bottom) simulations with transient and static internal temperatures. EnergyPlus-Polluto is not shown, as it overlaps entirely with the results from EnergyPlus-GCM.

EnergyPlus GCM and the Polluto model are essentially identical, with very small differences attributable to rounding. Discrepancies between the two EnergyPlus models and CONTAM are small when the same temperatures are used, and may be attributable to three factors: (1) the differences in calculating the wind pressures on the external surfaces of the buildings

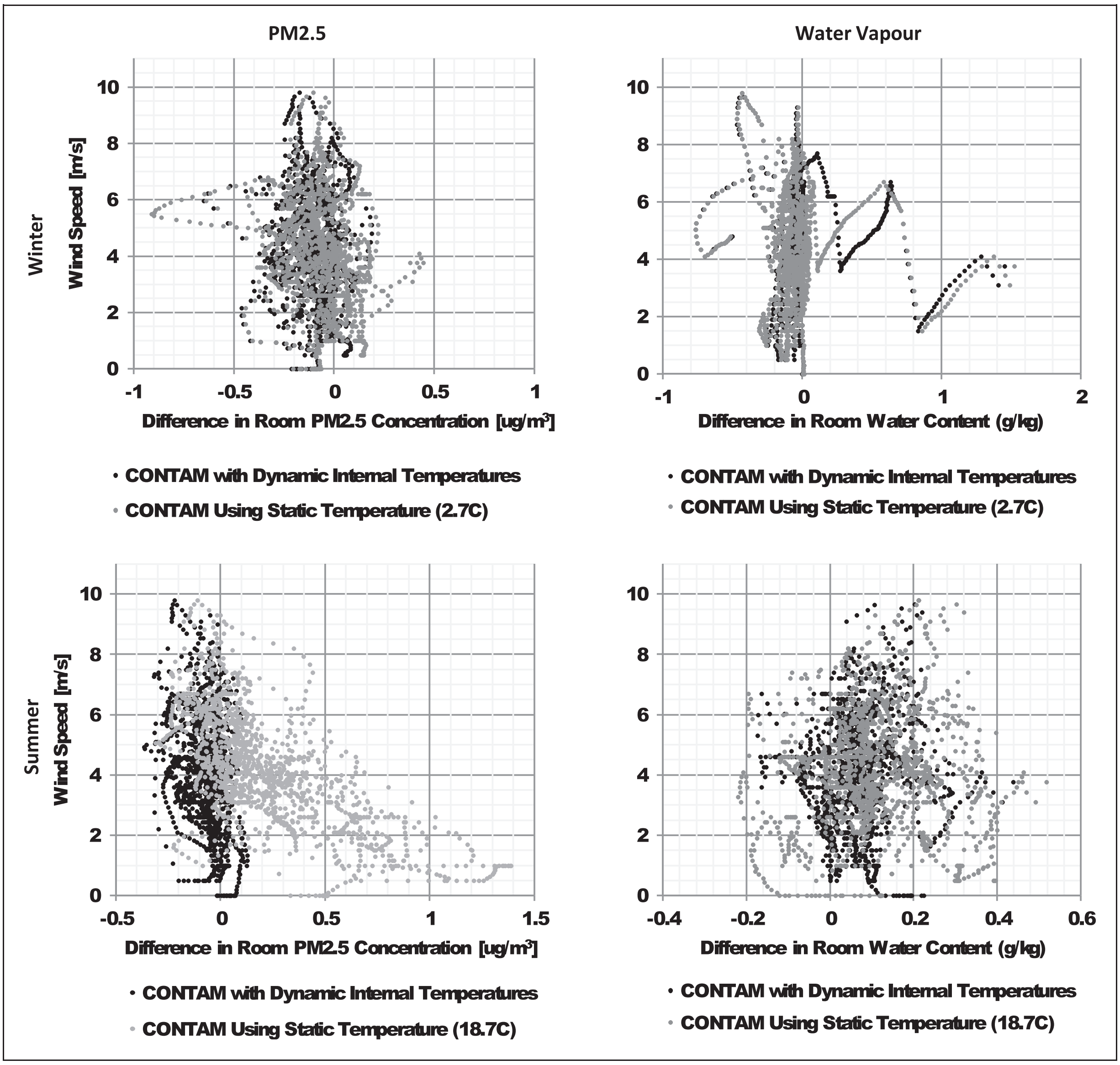

Figure 2. Differences in internal $PM_{2.5}$ (left) and water vapour content (right) for winter (top) and summer (bottom) simulation periods according to outdoor wind speed for transient EnergyPlus and static CONTAM models.

between the EnergyPlus and CONTAM models; (2) the lasting effects of the EnergyPlus WarmUp days, and (3) the instantaneous CONTAM output reporting versus the integrated EnergyPlus reporting methods. In CONTAM, non-trace contaminants are included in air density calculations; by treating moisture as a non-trace contaminant, any differences in the calculated density of the air should be minimised.

Differences in the predicted $PM_{2.5}$ concentration between the two models increase significantly when the internal temperatures are fixed in CONTAM rather than being derived from dynamic EnergyPlus calculations, demonstrating the importance of thermal performance of contaminant transport calculations. The differences were most significant when the constant temperature was set to the average outdoor temperature, and all thermal behaviour of the building envelope was ignored. At low wind speeds, the predicted indoor pollutant concentrations deviated by as much as 75% between models with dynamic and static internal temperatures.

The integration of a contaminant transport model into EnergyPlus gives a free fully-coupled thermal performance and contaminant transport model for buildings, eliminating the separate step of gathering internal temperature data for a CONTAM model for cases where the internal temperature is likely to fluctuate due to external weather conditions and building performance. In addition, EnergyPlus has the advantage of being coupled with a number of other modules that may impact contaminant transport. For example, the Heat and Moisture Transport (HAMT) model[18] can account for the hygrothermal behaviour of the building envelope, which can affect the internal water vapour concentration (non-trace contaminant) and subsequently the density of the air. Furthermore, another advantage of EnergyPlus is its ability to output integrated values rather than instantaneous values of pollutant concentration over time. In scenarios where short bursts of a pollutant can be generated, such as in cooking, CONTAM may fail to report elevated concentrations of pollutant if the time-step is not suitably short. In such scenarios, integrated output reporting saves writing-out time and memory as it is not necessary to output results with a small time-step. EnergyPlus GCM has the advantage over the Polluto model, in that it has been implemented in a more recent version of the EnergyPlus model, and can couple indoor contaminant levels with building performance – this feature could be used to model, for example, the influence of temperature-dependent window opening behaviour and the impact on indoor air quality. However, the EnergyPlus GCM is currently limited by being restricted to a single contaminant, while Polluto is capable of modelling many simultaneously.

While EnergyPlus may be a powerful tool for simulating the coupled thermal performance and airflow of a building, the model has a number of limitations in comparison to CONTAM and CONTAM-TRNSYS. The complexity of the EnergyPlus building simulation tool, and particularly the creation of an Airflow Network, may make the model inaccessible to non-expert users. In addition, CONTAM is able to model a number of mechanical ventilation systems, whereas the ability of EnergyPlus to model mechanical ventilation systems using the Airflow Network is limited. In reality, most buildings will have some degree of heating and cooling that will allow them to operate within a 'fixed' range of temperatures, thus limiting the impact of the thermal performance of the building on air and contaminant movement that may occur if the temperatures were floating.

Conclusions

The EnergyPlus GCM and Polluto models predict indoor $PM_{2.5}$ concentrations and water content ratios very similar to those of CONTAM when both tools are given exactly the same building description and external environment. The programs agree to better than 5.0% on a minute-by-minute comparison over a three-week summer period for winter and summer scenarios. When internal temperatures are fixed in CONTAM and allowed to float in EnergyPlus, significantly larger differences between the results are observed, indicating the importance of internal temperatures on airflow and contaminant transport within buildings. The results indicate that EnergyPlus GCM and Polluto are useful tools for calculating internal contaminant transport coupled to the thermal performance of the whole building system.

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Conflict of interest

None declared.

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RESEARCH

Home energy efficiency and radon related risk of lung cancer: modelling study

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Abstract

Objective To investigate the effect of reducing home ventilation as part of household energy efficiency measures on deaths from radon related lung cancer.

Design Modelling study.

Setting England.

Intervention Home energy efficiency interventions, motivated in part by targets for reducing greenhouse gases, which entail reduction in uncontrolled ventilation in keeping with good practice guidance.

Main outcome measures Modelled current and future distributions of indoor radon levels for the English housing stock and associated changes in life years due to lung cancer mortality, estimated using life tables.

Results Increasing the air tightness of dwellings (without compensatory purpose-provided ventilation) increased mean indoor radon concentrations by an estimated 56.6%, from 21.2 becquerels per cubic metre (Bq/m^3) to 33.2 Bq/m^3. After the lag in lung cancer onset, this would result in an additional annual burden of 4700 life years lost and (at peak) 278 deaths. The increases in radon levels for the millions of homes that would contribute most of the additional burden are below the threshold at which radon remediation measures are cost effective. Fitting extraction fans and trickle ventilators to restore ventilation will help offset the additional burden but only if the ventilation related energy efficiency gains are lost. Mechanical ventilation systems with heat recovery may lower radon levels and the risk of cancer while maintaining the advantage of energy efficiency for the most airtight dwellings but there is potential for a major adverse impact on health if such systems fail.

Conclusion Unless specific remediation is used, reducing the ventilation of dwellings will improve energy efficiency only at the expense of population wide adverse impact on indoor exposure to radon and risk of lung cancer. The implications of this and other consequences of changes to ventilation need to be carefully evaluated to ensure that the desirable health and environmental benefits of home energy efficiency are not compromised by avoidable negative impacts on indoor air quality.

Introduction

Through the 2008 Climate Change Act,[1] the UK government has enshrined in law targets for reducing emissions of greenhouse gases as its commitment towards global action on climate change: compared with 1990 a 34% reduction by 2020, 80% by 2050, and a recommended interim goal of 60% reduction by 2030.[2] A key target for such reduction is the housing sector,[3] for which substantial population wide changes are needed over the coming decades to improve energy efficiency, primarily through better insulation of the fabric (walls, roof, and floor) of dwellings and tighter control of ventilation.

While control of ventilation is good for energy efficiency, indoor temperatures in winter,[4] and protection against outdoor pollutants (notably airborne particles),[5] it has the potential to increase concentrations of pollutants arising from sources inside or underneath the home.[6 7] Notable among these is radon, a naturally occurring inert gas formed from the radioactive decay of elements of the uranium series, which seeps into homes through the floor, especially in areas with predisposing geology and soil type.[8] Radon is the second most important risk factor for lung cancer after smoking and may be responsible for 15 000 to 22 000 deaths from lung cancer each year in the United States,[9] 9% of deaths from lung cancer in European countries,

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Supplementary information

and around 1400 cases annually in the United Kingdom.[10 11] Radon is unique in the context of indoor air quality since it is a continuous source, which is therefore not responsive to the intermittent ventilation techniques that can be used to deal with other pollutants at the emission source (for instance, using extraction fans to remove cooking related particulates).

The housing energy efficiency strategy for England will entail an intervention affecting almost all of the 22.3 million dwellings, reducing ventilation rates and increasing radon levels on a population wide basis. This is an issue that has received relatively little attention despite the large scale of planned investments in housing. If these actions are carried out in an inappropriate manner there is potential for a substantial adverse impact on public health that will be embedded in the population for years. We carried out a modelling study to estimate the impact such a strategy may have on radon levels and associated lung cancer mortality.

Methods

The study entailed two main components: building physics modelling of current and future radon levels in the housing stock of England, and a health impact model for lung cancer mortality based on a life table method.

Mitigation scenarios

We modelled indoor radon levels for the present day and for four future scenarios representing a variety of plausible retrofitting strategies, which could be applied to the existing stock to help achieve reduction targets for carbon dioxide emissions. The four future scenarios were:

Scenario 1 (air tightness)—the air tightness of the housing stock is increased in line with (a realistic interpretation of) good practice guidance on reducing uncontrolled ventilation in dwellings to help achieve improvements in household energy efficiency.[12] The specified change for scenario 1 represents a reduction in permeability of dwellings ("air leakiness"), from the current average of 13 $m^3/m^2/h$ at 50 Pa pressure to 7 $m^3/m^2/h$, with a target upper limit for air permeability of 10 $m^3/m^2/h$ (maximum allowed for new builds under Part L of the Building Regulations for England[13] instead of the recommended "good practice maximum" of 5 $m^3/m^2/h$. Moreover, we assumed that 9% of dwellings fail to meet this target and are therefore above 10 $m^3/m^2/h$, a failure rate informed by empirical evidence on currently achieved permeability levels in refurbished[14] and new build dwellings.[15]

Scenario 2 (air tightness+purpose-provided ventilation)—as for scenario 1 but with the addition of partially compensating purpose-provided ventilation (trickle vents and extraction fans) in all dwellings to offset some of the reduction in air exchange. We assumed such purpose-provided ventilation was not used or was non-operational in 40% of dwellings.[16]

Scenario 3 (with mechanical ventilation and heat recovery)—as for scenario 2 but with mechanical ventilation and heat recovery systems installed in the 20% most airtight dwellings (permeability ≤3 $m^3/m^2/h$). Mechanical ventilation and heat recovery systems pump air through dwellings but recover heat from the expelled air, so maintaining relatively high air exchange but with the advantage of heat recovery to save energy. These systems are a potentially efficient solution for very airtight dwellings, the efficiency of which can be identified using a standard blower door test.

Scenario 4 (with mechanical ventilation and heat recovery assumed to include 10% failures)—as for scenario 3 but assuming that 10% of mechanical ventilation and heat recovery systems fail or are not used appropriately.

Modelling radon levels

For each of the present day and future stock scenarios, we modelled the distribution of indoor radon levels using the validated multizone model, CONTAM.[17] We modelled 10 housing archetypes (seven archetypes of houses and three of flats) under a range of ventilation strategies (purge (window opening) ventilation only or purge ventilation plus either trickle ventilators or extraction fans (in bathrooms and kitchens), or both) depending on dwelling type and age. We also modelled the inclusion of mechanical ventilation and heat recovery systems for the most airtight dwellings. Operational characteristics of extraction fans, trickle ventilators, and mechanical ventilation and heat recovery systems were matched to UK industry norms and specified to comply with minimum whole house ventilation rates required by Approved Document F of the Building Regulations for England and Wales.[18] We matched the present day (baseline) frequency of archetype and ventilation method combinations to data from the English Housing Survey 2009.[19] The distribution of air permeabilities in dwellings (see supplementary fig 1) was based on extensive survey measurements.[20] Figure 1⇓ shows the modelled ventilation rate (air changes per hour) distribution for each scenario.

We applied a radon emission rate to all dwellings proportional to the area of the ground floor rooms.[21] We assumed that flats on the first floor had 50% of the ground floor radon levels, whereas flats above the first floor were not affected by radon.[22] To account for geographical variations in radon levels, we constructed models for areas of low, medium, and high radon exposure by multiplying the modelled exposures by factors determined by calibration against observed data.[23]

Greenhouse gas emissions

We estimated the space heating demand of the stock due to ventilation heat losses using the standard degree hour method,[24 25] assuming a heating efficiency of 77%.[26] This was used to estimate the corresponding greenhouse gas emissions for England in megatonnes of carbon dioxide equivalent (Mt CO_2e) based on the current carbon intensity[27] and under decarbonisation assumptions consistent with the UK's 2020 and 2030 climate change mitigation targets.

Modelling impact on lung cancer mortality

We estimated the impact of altered radon levels on lung cancer mortality using life table methods based on the IOMLIFET model,[28] populated using age specific population data and 2009 rates for all cause and lung cancer specific mortality for England and Wales obtained from the UK Office for National Statistics. The model estimates patterns of survival in the population over time, with outputs including changes in the number of deaths and life years lived each year. To perform the health impact assessment, we adjusted the mortality rates in response to the changed exposures to radon and the outputs compared against those of the baseline (unadjusted) life tables. We modelled health impacts over a follow-up period of 106 years; long enough for the original birth cohort to have died out (105 was maximum age in life table). For the main analyses, we assumed no changes in the underlying health status of the population over time, which previous work has shown has only a minor effect on life table calculations.[29]

To make clearer the impact of changes in ventilation, we assumed an instantaneous step change in stock ventilation characteristics under each of the future scenarios. In reality, implementation would be phased over time. However, we did incorporate time dependent functions to model the latency between change in exposure and changes in lung cancer mortality. The assumed sigmoid onset lag for increased exposure assumed close to zero excess risk within 10 years of increased exposure and a gradual rise to almost full excess risk by 20 years. For reduced exposure, the assumed cessation lag was an exponential decline (see supplementary fig 2). In both cases, we applied a proportion of the relative risk each year after the intervention, with the full relative risk applied after 20 years.

We assumed a linear no threshold model for the relation between radon level and risk of lung cancer with a 16% increase in lung cancer mortality risk per 100 Bq/m^3 based on evidence from European case-control studies.[10 30] This relation has been corroborated by other studies and meta-analyses[31 32] and is consistent with evidence that radon is a likely carcinogen at all exposure levels.[33]

As smokers have a greatly increased risk of lung cancer (although their radon related risk is proportionate in relative terms to that of non-smokers),[10 34] we used separate life tables for smokers and non-smokers, assuming lung cancer rates in smokers to be 25 times that of non-smokers.[10] Information on the current smoking prevalence in England (21% in 2009) was based on survey data.[35] In the base case scenario, we assumed a 50% decrease in lung cancer prevalence to account for the lagged effect of the roughly 50% decrease in smoking in the past 30 years on future underlying lung cancer mortality rates, but no further decreases in lung cancer rates owing to possible further future reductions in smoking. However, in sensitivity analyses, we did examine the effect of lower future smoking prevalence (of 15% and 10%) as well as of removing the lagged effect of the recent decline in smoking prevalence. We did not model synergistic effects of environmental (second hand) tobacco smoke on lung cancer risk as presently evidence allowing accurate quantification of such impacts is insufficient.

Results

Radon levels

We calibrated our model based estimates of current radon levels to approximate the observed distribution for England and Wales (modelled mean 21.2 Bq/m^3, mean from survey data 21.0 Bq/m^3) (see supplementary fig 3).[23 36] Table 1⇓ summarises the radon levels under present day and each of the four future scenarios (see also supplementary fig 4). With the increased air tightness of scenario 1, radon levels increase by 56.6% from the present day mean of 21.2 Bq/m^3, to 33.2 Bq/m^3. A substantial increase also occurs in the proportion of dwellings above the Public Health England Action Level of 200 Bq/m^3.[34] The increase from 0.6% to 2.0% would represent a further three quarters of a million people living in homes with radon above the Action Level.

In scenario 2, the addition of purpose-provided ventilation (assumed to operate correctly in 60% of homes) reduces the increased radon levels but does not restore them to present day levels. However, models that (unrealistically) assume 100% operation for purpose-provided ventilation in fact reduce radon to marginally below current levels (data not shown).

Assuming mechanical ventilation and heat recovery is installed in the 20% most airtight dwellings (scenario 3) has a considerable impact, reducing both the number of homes with the highest levels of radon and the population mean to 19.6 Bq/m^3, slightly below current day levels.

Assuming a 10% failure in mechanical ventilation and heat recovery systems (scenario 4) results in only a modest increase in the mean, to 21.8 Bq/m^3, because the failure affects only 2% of the housing stock (10% of the 20% with mechanical ventilation and heat recovery). However, people in homes with failure of mechanical ventilation and heat recovery systems would experience substantial increases in radon levels, of more than 1000 Bq/m^3 in some circumstances, although it is likely that many homeowners would eventually fix such faulty systems or adjust their behaviour (for example, by opening windows more often) to improve air exchange.

Health impacts and greenhouse gas emissions

Translation of our modelled distribution of present day radon levels into risk of lung cancer mortality suggests that current levels account for around 1000 deaths per year in England, a figure slightly lower than published estimates.[11 23] More than 90% of this lung cancer burden from radon relates to levels below 200 Bq/m^3, and over 40% to levels below 24 Bq/m^3 (fig 2⇓).

The 12.0 Bq/m^3 increase in mean indoor level under scenario 1 was estimated to increase the attributable burden of lung cancer mortality by a peak of around 4700 life years lost and 278 additional deaths per year. Over the 106 year follow-up period, 367 200 fewer life years would be lived by the population, representing about 3500 life years lost per year on average. These impacts would, however, vary over time (table 2⇓). Changes in life years lost in the population would be negligible in the first decade or so after the intervention owing to the lag in lung cancer onset (fig 3⇓) and then increase rapidly, reaching a (sustained) peak after around 40 years and remaining relatively constant thereafter. Mortality impacts would be felt differently in different age groups (fig 4⇓), with the increase in radon related deaths at younger ages reducing the size of the population (and so the number of deaths) in older age. Over the long term, the net effect would be a shift towards deaths at younger ages and a decrease in life expectancy. The average reduction in ventilation related carbon dioxide equivalent emissions for England for this scenario was estimated to be 5.7 Mt CO_2e annually based on the emissions intensity for the current energy supply mix, or 2.3 Mt CO_2e with the energy mix expected by 2030 if the 60% target reduction in carbon intensity is achieved (table 2).

The addition of appropriate purpose-provided ventilation under scenario 2, which mitigates the increase in radon levels, was estimated to be associated with a peak annual radon related lung cancer burden of around 100 additional deaths and almost 1700 life years lost, with 130 900 life years lost over the follow-up period. Savings in carbon dioxide equivalent emissions were correspondingly smaller than in scenario 1. Benefits to health and to carbon emissions were found by incorporating mechanical ventilation and heat recovery in the most airtight dwellings (scenario 3), although scenario 4 shows the importance of ensuring these systems are functioning correctly.

Figure 5⇓ illustrates the trade-off between decreasing ventilation for improved energy efficiency and impact on radon related lung cancer mortality. To maximise ventilation related energy efficiency requires moving dwellings towards the left of the graph where ventilation and hence heat losses are low. However, as the plots for different house archetypes show, exposure to radon increases.[6 37] The shape of the curves indicates a particularly steep rise in the radon burden as ventilation rates approach very low levels below about 0.3 air changes per hour. The trade-off is shown explicitly in the lower plots of fig 5, with radon exposure translated into annual health burden (ignoring the onset time lag) and space heating demand translated into annual greenhouse gas emissions.

Sensitivity analysis

Assumptions about a potentially lower future prevalence of smoking (15% and 10%) indicate that any future radon related adverse health impacts could be smaller than suggested by the estimates presented here, which assume persistence of current smoking rates (table 3⇓). However, assuming no lagged effect of past reductions in smoking prevalence (that is, current lung cancer rates) would increase the impacts presented here. The results indicate that reduction in smoking is a potentially effective strategy for reducing much of the current burden from radon related lung cancer. However, such reductions are not guaranteed, whereas the increases in indoor radon levels are fixed until such time as other interventions are put in place to improve ventilation. In addition, decarbonisation of the energy mix for household energy would progressively erode the benefit of a reduction in ventilation related carbon dioxide equivalent emissions (table 2).

Discussion

This study suggests that energy efficiency interventions that increase the air tightness of dwellings without compensatory purpose-provided ventilation will increase indoor radon concentrations and associated lung cancer risks. The reduced air exchange accompanying efficiency upgrades that meet 2030 GHG abatement targets is likely to increase radon levels by over 50% with an additional annual health burden of close to 5000 life years lost from lung cancer, albeit with a delayed evolution because of the latency of disease. Moreover, fitting extraction fans and trickle ventilators to restore ventilation will help offset the additional burden only if the ventilation related energy efficiency gains are lost. In other words, leaving aside the use of mechanical ventilation and heat recovery, ventilation related improvements in energy efficiency can be achieved only at the expense of additional radon related lung cancer burdens unless there is widespread use of remediation.

Although trends in radon related health burdens may be helped if effective action is taken to reduce smoking prevalence over coming decades, the relative benefit of reduced ventilation on carbon dioxide equivalent emissions is likely to decline over time with progressive decarbonisation of household energy supplies. Even with today's relatively "leaky" housing stock, ventilation related heat losses account for a comparatively modest fraction (around 15%) of all dwelling heat losses(equivalent to around 13 Mt CO_2e of the UK's 600 Mt CO_2e total emissions).[38] Thus the ratio of the positive effects on carbon dioxide equivalent emissions against the detrimental effects on radon related lung cancer will almost certainly become less favourable over time unless clinical treatments become noticeably more effective (which is possible). In addition, our modelling of measures to reduce ventilation under scenario 1 reduces space heating demand for ventilation by 34% (table 2), consistent with 2020 abatement targets, but only half of that needed to achieve 2030 targets: a proportionate reduction in air exchange for the 2030 target would imply substantially greater increases in radon levels and hence risk to health.

Caution is therefore needed to ensure that risks from radon are minimised by appropriate compensatory ventilation systems or cost effective remediation measures. However, a particular challenge for health protection is that the additional burden of radon related deaths from lung cancer is not concentrated in homes with radon above the UK Action Level of 200 Bq/m^3 or even the Target Level of 100 Bq/m^3. Rather, the bulk of additional radon deaths would arise in the millions of homes exposed to levels of radon well below those where conventional remediation is considered cost effective (fig 2).[23 39 40] This is an example of what Rose has called the prevention paradox.[41] Given the (assumed) linear no threshold relation between radon level and lung cancer,[10 30] any upward shift of indoor radon levels across most dwellings has the potential for a genuinely adverse impact at population level; and the same would apply to any other pollutant of indoor origin.

Our evidence also suggests that adding mechanical ventilation and heat recovery in the most airtight dwellings may appreciably reduce indoor radon levels. However, it can only be introduced in the most airtight dwellings (and few current dwellings come close to the required levels of air tightness), pressure differentials may in some circumstances exacerbate radon levels,[42] and, as yet, experience with it has been insufficient to know how well it would work in practice over the long term. Failure of mechanical ventilation and heat recovery systems (through incorrect installation, operation, maintenance, or use) could result in extremely high levels of radon.

Strengths and limitations of this study

The strength of this study has been the ability to combine detailed models of the housing stock, radon levels, and population health to assess a major area of government strategy planned for the coming decades. It is the first study of its kind to model future radon levels and health impacts under climate change mitigation scenarios in such detail and to study the distribution of impacts across the entire housing stock. The models are, of course, somewhat artificial constructs that can never provide entirely accurate representations of such a complex system, and many uncertainties exist. For the purposes of this study we have assumed that people are static. Although individual exposures could change as people relocate, at the population level this should not affect the modelled exposures and health impacts as one household is generally replaced by another: some people may move to more polluted dwellings, whereas others may move to less polluted ones, but the average change in risk of lung cancer remains unaffected. We have incorporated typical occupant behaviour schedules in our models and assumed no changes in behaviour subsequent to the introduction of new technologies. Behaviours will mean some variation in indoor radon levels from dwelling to dwelling (all other things held constant), but our model reflects the current (empirical) distribution of levels, and we consider it reasonable to assume no major change in behaviour from today. Certainly there is little evidence from which to conclude that there would be any change. If future decreases in smoking prevalence are substantial, this could help to ameliorate the adverse impact of increased radon levels, as shown by the sensitivity analyses. Although this provides further reason to encourage smoking cessation, assumption of possible success in smoking reduction is no justification for allowing radon levels to rise. Moreover, decreased ventilation in dwellings will possibly increase second hand exposure to tobacco smoke in households with smokers, a factor that has not been taken into account in our estimations of burden. Finally, we have also not included the full spectrum of potential radon related health outcomes, such as leukaemia,[43] since presently evidence to permit quantification of such impacts is insufficient.

Comparison with other studies

Although uncertainties exist, our model is almost definitely correct about the general direction of change, as the physics dictate that lower air exchange means higher levels of radon,[44] and correct also that energy efficiency achieved by reduced ventilation will result in higher radon related health burdens unless there is specific remediation.[23] Moreover, our estimates of the magnitude of changes in radon levels are broadly in line with previous modelling work,[45 46 47 48] which, as the Swiss Federal Office for Public Health notes, also suggests the potential for a "frequent, sometimes drastic increase" in radon levels after energy efficiency interventions.[49]

Conclusions and policy implications

Our results have important implications for current UK policy related to housing energy efficiency. They should not be interpreted as providing evidence against the desirability of improving home energy efficiency in general. However, reducing ventilation as part of these measures will embed changes for millions of dwellings that may carry substantial detrimental (as well as positive) effects on health while making only a modest contribution to energy efficiency. There is therefore a need for a more careful re-evaluation of how retrofitting of dwellings is carried out to ensure that the potential benefits, including those to health, are not compromised by injudicious air tightening.[50 51] There are different ways of achieving the same end: with regard to radon, a safer strategy might be to place greater emphasis on other measures to reduce energy use, such as improving the conduction properties of dwellings (insulation) and the decarbonisation of the energy supply.

Increasing the energy efficiency of housing is still likely to be a net benefit for health in many cases. This work does not challenge the view that there are generally good reasons for seeking to improve the energy efficiency of housing in England and in many other settings for health as well as for environmental reasons.[45] The caution is in how those energy efficiency improvements are implemented. Radon is just one of several environmental exposures that may be altered by increasing the air tightness of dwellings, some of which, including second hand tobacco smoke and particles of indoor origin, may be adversely affected, whereas others, including indoor temperatures in winter, may be improved.[52] Optimising ventilation strategies for health is therefore more complex if all relevant exposures are taken into account.[53] However, our work highlights the potential problems that may be caused by energy efficiency measures that target heat losses from uncontrolled ventilation. This is a problem that needs much research and debate before undertaking the planned large scale programme of housing investments that may embed health problems for many years to come. For radon at least, caution is needed to ensure that the pursuit of energy efficiency does not precipitate an unwelcome increase in disease burden in the population as a whole. It is also a reminder that all forms of mitigation action have the potential for negative as well as for positive health impacts at population level and need to be carefully planned.

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Competing interests: All authors have completed the ICMJE uniform disclosure form at www.icmje.org/coi_disclosure.pdf (available on request from the corresponding author) and declare: no support from any organisation for the submitted work; no financial relationships with any organisations that might have an interest in the submitted work in the previous three years, no other relationships or activities that could appear to have influenced the submitted work.

Ethical approval: Not required.

Data sharing: No additional data available.

Transparency: The lead author affirms that the manuscript is an honest, accurate, and transparent account of the study being reported; that no important aspects of the study have been omitted; and that any discrepancies from the study as planned have been explained.

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What is already known on this topic

Radon is a radioactive inert gas that enters homes by seepage from the ground

It is the second most important risk factor for lung cancer after smoking and may be responsible for around 1400 cases annually in the United Kingdom

Major improvements to home insulation are expected to reduce energy use and meet climate change mitigation targets

What this study adds

Proposed strategies for reducing greenhouse gas emissions from the housing sector entail interventions that reduce uncontrolled ventilation, which are likely to increase indoor radon levels and associated lung cancer risk

The post-intervention increases in radon for the majority of homes that would contribute most of the additional lung cancer burden are below the threshold at which conventional radon remediation measures are cost effective

The implications of ventilation control on indoor radon levels need to be carefully evaluated before the roll-out of national schemes for improving home energy efficiency

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Tables

Table 1| Summary statistics of indoor radon concentrations for all scenarios

Scenarios	Radon concentration (Bq/m^3)			Percentage >200 Bq/m^3
	Mean	Median	95th centile	
Present (baseline)	21.2	12.5	73.3	0.6
Scenario 1 (air tightness)	33.2	19.5	121.2	2.0
Scenario 2 (air tighteness+purpose-provided ventilation)	25.5	13.9	94.6	1.2
Scenario 3 (as for scenario 2+MVHR)	19.6	11.1	69.8	0.5
Scenario 4 (as for scenario 3+10% failures in MVHR)	21.8	11.8	85.3	0.6

MVHR=mechanical ventilation and heat recovery systems.

Table 2| Modelled health impacts and estimated changes in stock annual space heating demand and greenhouse gas (GHG) emissions for different assumptions of decarbonisation of space heating energy supply

Scenarios	Change in life years lived by population*			Change in stock annual space heating demand for ventilation (TWh)	Change in stock annual GHG emissions (Mt CO_2e)†		
	0-20 years	0-50 years	Over follow-up period		No further decarbonisation	Assuming 34% decarbonisation (2020 target)	Assuming 60% decarbonisation (2030 recommended target)
Scenario 1 (air tightness)	–5200	–121 000	–367 200	–27	–5.6	–3.7	–2.2
Scenario 2 (air tightness+purpose-provided ventilation)	–1800	–43 100	–130 900	–15	–3.2	–2.1	–1.3
Scenario 3 (as for scenario 2+MVHR)	4000	21 500	54 000	–22	–4.5	–3.0	–1.8
Scenario 4 (as for scenario 3+10% failures in MVHR)	–300	–7000	–21 300	–22	–4.5	–3.0	–1.8

Mt CO_2e=megatonnes of carbon dioxide equivalent; TWh=terawatt hour; g/kWh=grammes per kilowatt hour; MVHR=mechanical ventilation and heat recovery systems.

*Figures rounded to nearest 100; negative figures indicate loss of life years.

†Assuming current carbon intensity of 208 g/kWh(38).

Table 3| **Sensitivity of health impacts to smoking prevalence and lung cancer mortality rate**

	Health impact (change in life years over follow-up period)*					
	Current (2009) lung cancer mortality			50% reduction from current (2009) lung cancer mortality		
	Assumed smoking prevalence			Assumed smoking prevalence		
Scenarios	**21% (current)**	**15%**	**10%**	**21% (current) (base case)**	**15%**	**10%**
Scenario 1 (air tightness)	–733 800	–558 700	–412 900	–367 200	–279 600	–206 600
Scenario 2 (air tightness+purpose-provided ventilation)	–261 700	–199 300	–147 200	–130 900	–99 700	–73 600
Scenario 3 (as for scenario 2+MVHR)	108 100	82 300	60 800	54 000	41 100	30 400
Scenario 4 (as for scenario 3+10% failures in MVHR)	–42 500	–32 400	–23 900	–21 300	–16 200	–12 000
Approximate % change in health impact relative to base case	100	52	12	0 (base case)	–24	–44

MVHR=mechanical ventilation and heat recovery systems.

*Figures rounded to nearest 100; negative figures indicate loss of life years.

Figures

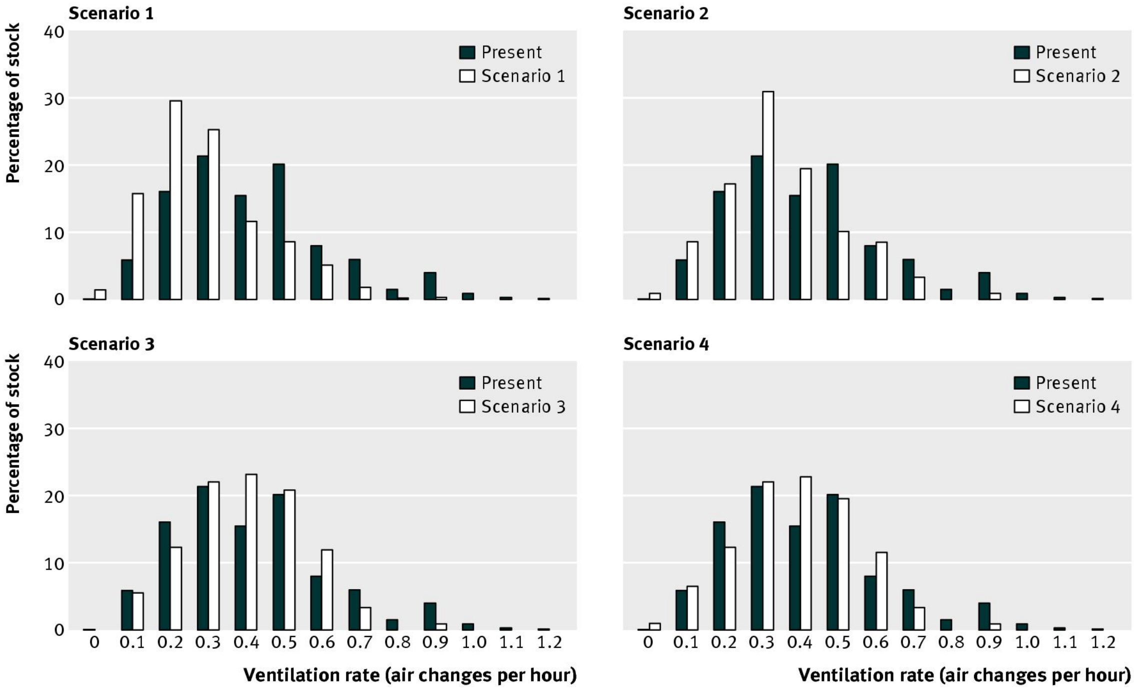

Fig 1 Modelled present day and future ventilation rate distributions of English housing stock. Scenario 1=air tightness; scenario 2=air tightness+purpose-provided ventilation; scenario 3=as for scenario 2+mechanical ventilation and heating recovery (MVHR); scenario 4=as for scenario 3+10% failures in MVHR

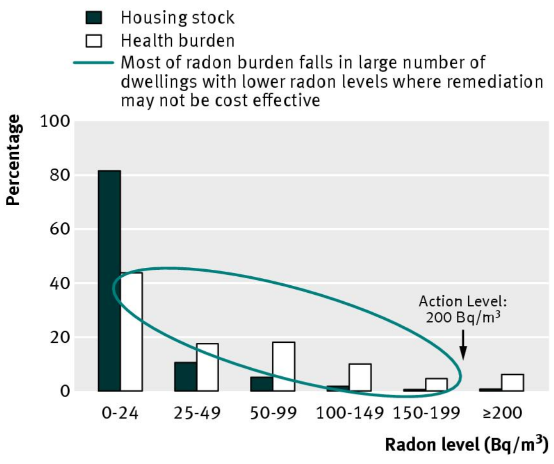

Fig 2 Proportions of current housing stock and attributable burden of radon related lung cancer mortality for different levels of radon

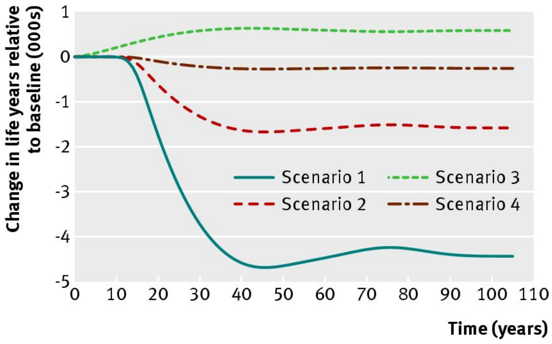

Fig 3 Change in life years lived in population (relative to baseline) over time for each scenario. Negative figures indicate loss of life years. Scenario 1=air tightness; scenario 2=air tightness+purpose-provided ventilation; scenario 3=as for scenario 2+mechanical ventilation and heat recovery (MVHR); scenario 4=as for scenario 3+10% failures in MVHR

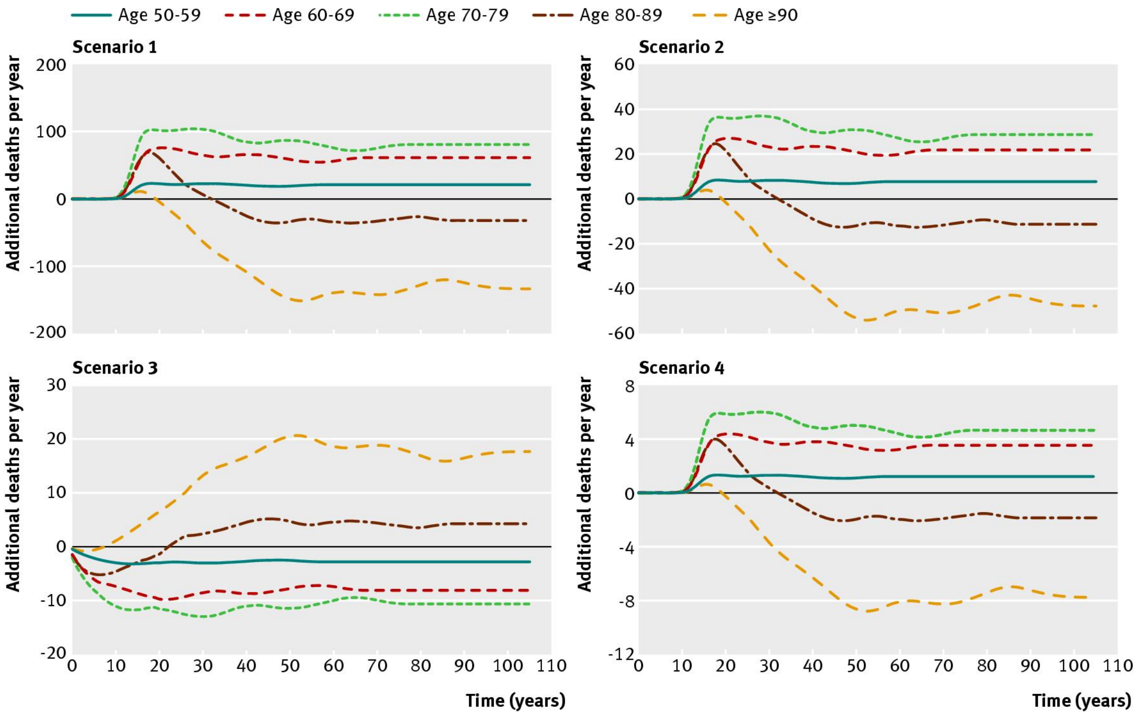

Fig 4 Additional deaths per year (relative to baseline) over time for each scenario and for different age groups. Scenario 1=air tightness; scenario 2=air tightness+purpose-provided ventilation; scenario 3=as for scenario 2+mechanical ventilation and heat recovery (MVHR); scenario 4=as for scenario 3+10% failures in MVHR. Note changes of scale on y axes

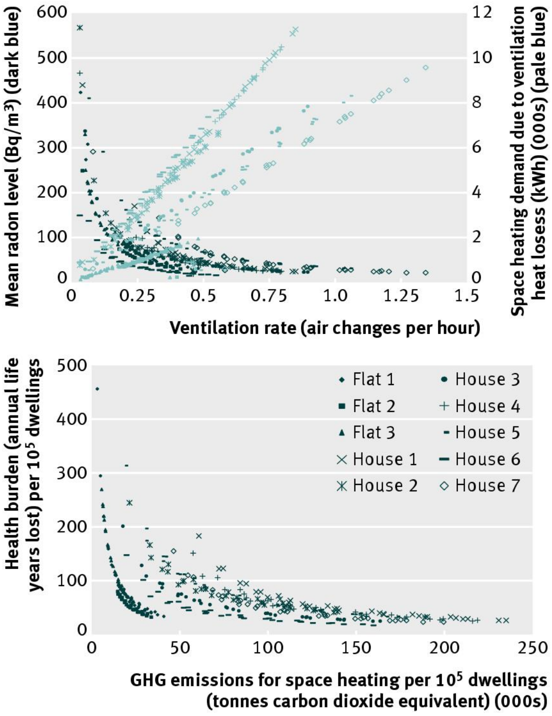

Fig 5 Mean radon level and space heating demand due to ventilation heat losses for the English housing stock plotted against ventilation rate, and current attributable health burden (annual life years lost assuming no lag) compared with annual greenhouse gas (GHG) emissions for space heating per 10^5 dwellings

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The modifying effect of the building envelope on population exposure to $PM_{2.5}$ from outdoor sources

Abstract A number of studies have estimated population exposure to $PM_{2.5}$ by examining modeled or measured outdoor $PM_{2.5}$ levels. However, few have taken into account the mediating effects of building characteristics on the ingress of $PM_{2.5}$ from *outdoor* sources and its impact on population exposure in the *indoor* domestic environment. This study describes how building simulation can be used to determine the indoor concentration of outdoor-sourced pollution for different housing typologies and how the results can be mapped using building stock models and Geographical Information Systems software to demonstrate the modifying effect of dwellings on occupant exposure to $PM_{2.5}$ across London. Building archetypes broadly representative of those in the Greater London Authority were simulated for pollution infiltration using EnergyPlus. In addition, the influence of occupant behavior on indoor levels of $PM_{2.5}$ from outdoor sources was examined using a temperature-dependent window-opening scenario. Results demonstrate a range of I/O ratios of $PM_{2.5}$, with detached and semi-detached dwellings most vulnerable to high levels of infiltration. When the results are mapped, central London shows lower I/O ratios of $PM_{2.5}$ compared with outer London, an apparent inversion of exposure most likely caused by the prevalence of flats rather than detached or semi-detached properties.

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Practical Implications

Population exposure to air pollution is typically evaluated using the outdoor concentration of pollutants and does not account for the fact that people in London spend over 80% of their time indoors. In this article, building simulation is used to model the infiltration of outdoor $PM_{2.5}$ into the domestic indoor environment for dwellings in a London building stock model, and the results mapped. The results show the variation in relative vulnerability of dwellings to pollution infiltration, as well as an estimated absolute indoor concentration across the Greater London Authority (GLA) scaled by local outdoor levels. The practical application of this work is a better understanding of the modifying effect of the building geometry and envelope design on pollution exposure, and how the London building stock may alter exposure. The results will be used to inform population exposure to $PM_{2.5}$ in future environmental epidemiological studies.

Introduction

Due to high volumes of traffic, a dense road network, and proximity to major traffic hubs such as Heathrow, London experiences a high level of outdoor $PM_{2.5}$ air pollution relative to the rest of the UK. Population exposure to $PM_{2.5}$ has been associated with negative health effects. In England, the fraction of mortality attributable to anthropogenic particulate air pollution in 2011 is estimated to be 5.36%, while in Greater

London it is 7.17% (PHE, 2013). Earlier studies have estimated that in London in 2008, $PM_{2.5}$ caused mortality equivalent to around 4000 deaths and that a permanent 1 $\mu g/m^3$ reduction in $PM_{2.5}$ would add 400 000 years of life for the current population (Miller, 2010). Internationally, $PM_{2.5}$ is estimated to cause about 3% of all mortality from cardiopulmonary disease, about 5% of mortality from cancer of the trachea, bronchus, and lung, and around 1% of mortality from acute respiratory infections in children under 5 years old (Cohen et al., 2005). While the total $PM_{2.5}$ emissions in the UK are predicted to decrease by 25% by 2020 relative to 2009 levels, there is no known 'safe' level of $PM_{2.5}$ and there will continue to be health risks associated with exposure (DEFRA, 2013).

A number of studies have examined the epidemiological relationship between exposure to pollution and negative health effects [for example, Atkinson et al. (2013) and Tonne and Wilkinson (2013)]. However, these studies focus on outdoor pollution concentrations and population health and do not account for pollution in the indoor environment. Individuals in developed countries spend the majority of their time indoors; a study of pollution exposure in different microenvironments in London found participants were spending 80% of their time indoors, with 48–53% of their time spent in their homes during summer and winter, respectively (Kornartit et al., 2010). Therefore, the *indoor* pollution levels have a significant influence on an individual's exposure to pollution, and a building's airtightness and the manner in which it is operated can have a major impact on pollution ingress from the outdoor environment. Epidemiological studies typically use pollution measurements from urban background monitoring stations or modeled outdoor pollutant concentrations to estimate exposure; however, this may not offer a true representation of the exposure to a population spending time largely indoors. Indeed, a study of population exposure to $PM_{2.5}$ in different microenvironments found a good correlation between residential indoor levels and personal exposures (Lai et al., 2004).

$PM_{2.5}$ infiltration into buildings from external sources will depend on a number of factors, including the location, height, orientation, sheltering, and permeability of the building envelope, building geometry, the ventilation systems of the building, weather and urban meteorology conditions such as urban street canyons, and building occupant practices such as window opening and heating. In addition to infiltration, concentrations of $PM_{2.5}$ in dwellings will be affected by emissions from indoor sources such as cooking, smoking, as well as general domestic activities such as cleaning, dusting, and showering (Shrubsole et al., 2012). Removal of $PM_{2.5}$ from indoor and outdoor sources from the indoor air can occur through exfiltration, deposition onto surfaces, and filtration using mechanical ventilation systems.

Examining the relationship between indoor and outdoor pollution levels can be performed using field measurements or through modeling approaches. A number of studies have monitored the indoor concentration of $PM_{2.5}$ in different countries [see Chen and Zhao (2011), for a comprehensive review]. In the UK, there have been field studies measuring indoor $PM_{2.5}$ in dwellings with roadside, urban, and rural measurements (Jones et al., 2000) and distance to major roads (Kingham et al., 2000); studies comparing indoor UK levels to other European cities (Hoek et al., 2008; Lai et al., 2006); seasonal variations in indoor $PM_{2.5}$ exposure concentrations (Mohammadyan, 2005; Wheeler et al., 2013); short-term temporal variations associated with indoor activities (Gee et al., 2002; Wigzell et al., 2013); and in the homes of individuals with respiratory illnesses (Osman et al., 2007). A comparison between different building types by Nasir and Colbeck (2013) monitored indoor $PM_{2.5}$ levels in three different types of dwelling and found differences between them; however, differing occupant practices make it difficult to isolate the influence of the building on indoor $PM_{2.5}$ levels.

Modeling methods have also been used to characterize the indoor concentration of $PM_{2.5}$. Multizone mass transport models can be used to calculate concentration levels in buildings for exposure assessments (Milner et al., 2011). Studies examining the influence of ventilation and filtration interventions (Emmerich et al., 2005) and energy efficiency interventions (Das et al., 2013) on indoor $PM_{2.5}$ concentrations have been performed using the CONTAM modeling tool. The indoor $PM_{2.5}$ concentration across a building stock has been modeled using CONTAM for dwellings based on Boston public housing developments (Fabian et al., 2012) and the impact of energy efficient refurbishments in London's domestic stock (Shrubsole et al., 2012).

While a number of studies have examined the relationship between indoor and outdoor air pollution levels in dwellings, there has been little research on how the relative infiltration of a geographically distributed building stock can modify pollution exposure across an urban area. Chen et al. (2012) used typical infiltration rates of dwellings in US cities to estimate indoor exposure to particulate matter in a study examining short-term mortality rates; however, this study did not examine more local variations in building types and pollution levels. Furthermore, existing infiltration modeling approaches have estimated ventilation according to a schedule of activities without coupling behavior to indoor conditions such as temperature. The research presented here examines how the characteristics and geographical distribution of residential building types in the London building stock may affect the exposure levels of dwelling occupants to $PM_{2.5}$ from external sources. The whole-building simulation tool

EnergyPlus 8.0 (US-DOE, 2013) was used to model the infiltration of $PM_{2.5}$ into the indoor environment for dwellings broadly representative of the Greater London area (GLA) building stock. Two different scenarios were considered to demonstrate the influences of the building envelope and occupant practices: (i) pollution infiltration through cracks in the building fabric only and (ii) infiltration through cracks and temperature-dependent window opening. In both cases, trickle vents were included where appropriate, while extract fans were excluded due to their intermittent use and their assumed small contribution to time-averaged indoor concentrations of outdoor $PM_{2.5}$. Other mechanical ventilation systems, such as mechanical ventilation heat recovery (MVHR) or air conditioning (AC), were ignored due to their rarity in the UK domestic stock. The simulation results were used to develop indoor/outdoor (I/O) ratios describing the relationship between outdoor and indoor concentration of $PM_{2.5}$. These functions were then applied to calculate the indoor concentrations of outdoor-sourced $PM_{2.5}$ based on mapped external concentrations, temporal variations, and geographical location of dwelling types. The results were combined with existing $PM_{2.5}$ pollution maps to understand how dwellings may affect population exposure to particulate air pollution.

Method

The research area selected was the GLA, an area encompassing the 32 boroughs of London (Figure 1). The area has good mapped coverage of building data and has been the focus of both measured and modeled studies of $PM_{2.5}$ in outdoor air, with data on observed or estimated outdoor $PM_{2.5}$ levels available from a range of sources. The different inputs required for the model and how they relate to the project workflow can be seen in Figure 2. While the London population spends a significant amount of their time inside offices or buildings that are not their homes, commercial buildings can have significantly different indoor pollution levels due to HVAC system operation, filters, and complex building geometries. Spatial and archetype information on the commercial building stock is not widely available, and thus, this study focuses only on dwellings.

Building archetypes

A total of 15 dwelling archetypes developed for studies into overheating risk in London were used as a basis for the EnergyPlus modeling of $PM_{2.5}$ penetration through the building envelope (Oikonomou et al., 2012). This English Housing Survey (EHS) (DCLG, 2008), derived archetypes, with unique built form/age classifications, represents 76% of the known dwelling stock in the GLA according to the Geoinformation Group (GG) Building Class Geodatabase (GG, 2013).

Building fabrics were modeled with *U*-values derived for the building archetypes using the Standard Assessment Procedure for Energy Rating of Dwellings (SAP) (BRE, 2009), with the assumption that buildings have

Fig. 1 Research area: Greater London. Areas without dwelling information are shown in gray

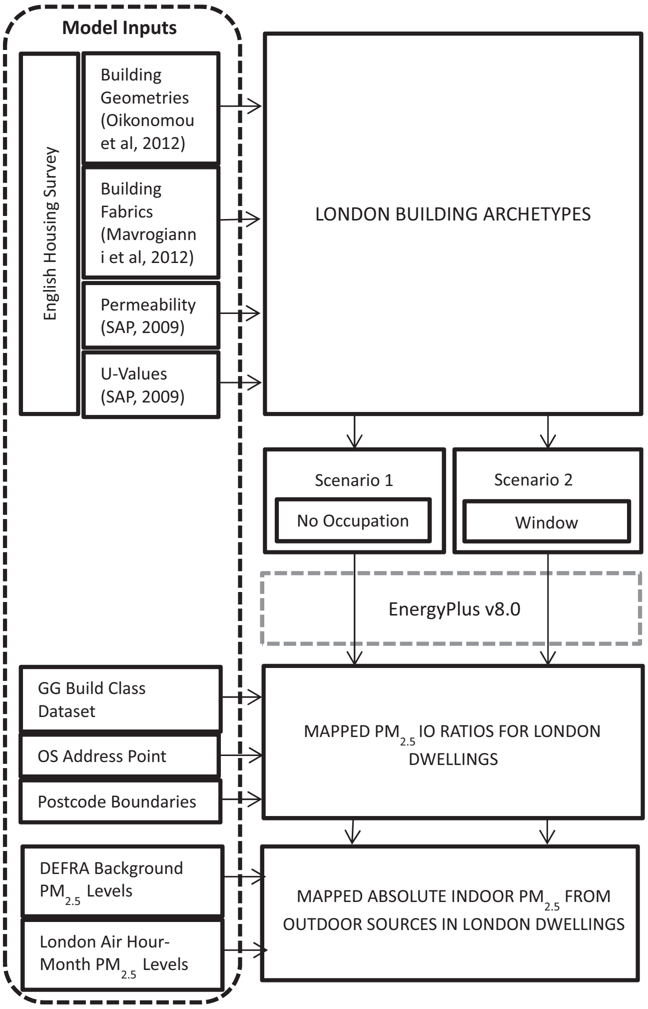

Fig. 2 Research workflow and data inputs

the most frequently occurring building fabric types according to the EHS. The building fabric can influence the indoor temperature in dwellings (Mavrogianni et al., 2012), which may lead to changes in the window opening behavior of the building occupants. Internal temperatures can also have an influence on airflow dynamics at low wind speeds due to stack effects. However, the overall effect of variations in insulation levels on yearly average indoor $PM_{2.5}$ levels is expected to be small, and so potential retrofits were ignored. Fabric *U*-values for the building archetypes can be seen in Table S1.

The permeability of the building archetypes was determined using the methodology in the SAP documentation (BRE, 2009), which accounts for infiltration through chimneys and vents, walls, floors, and windows and increased infiltration in multistorey buildings. Infiltration rates were calculated for each building in the EHS database (with and without any reduction to the rate caused by party walls), and the mean for each built form/age classification of the archetypes in the study determined. The infiltration rates were then converted to a permeability using the 'rule of 20' specified in the SAP methodology. The distribution of the estimated permeabilities in the EHS was compared with that of a field measurement study (Stephen, 2000), with good results.

Simulations were run with and without the presence of trickle vents, and the results weighted according to the estimated prevalence of the vents across the UK building stock [all dwellings post-1990, and 5% of pre-1990 dwellings (DECC, 2011)], assumed to be the same as in London. A description of the building archetypes and estimated permeability can be seen in Table 1.

Table 1 Dwelling archetype descriptions and permeability estimated from the EHS and SAP

Archetype code	Dwelling archetype	Age bracket	Frequency in stock, %	Estimated permeability ($m^3/h/m^2$ at 50 Pa)
H01	Late Victorian/Edwardian Terrace (Large T)	1902–1913	15.4	17.2
H02	WW1 & WW2 Simple Terrace	1914–1945	14.5	14.9
H03	WW1 & WW2 Large Semidetached	1914–1945	8.8	16.1
H04	'60s & '70s Tall Purpose-built Flats	1960–1979	5.7	16.2
H05	Late Victorian/Edwardian Simple Terrace	1902–1913	5.5	17.2
H06	Post-War Tall Purpose-built Flats	1946–1959	4.7	13.2
H07	Recent Tall Purpose-built Flats	1980–2008	3.6	9.2
H08	Late Victorian/Edwardian Simple Terrace (attic)	1902–1913	2.9	17.2
H09	WW1 & WW2 Bungalow	1914–1945	2.4	17.9
H10	'60s & '70s Simple Terrace	1960–1979	2.4	12.3
H11	'60s & '70s Line-built Walk-up Flats	1960–1979	2.3	9.8
H12	WW1 & WW2 Line-built Walk-up Flats	1914–1945	2.1	10.1
H13	Recent Terrace with Shop Below	1980–2008	2.1	12.2
H14	Post-War Step-Linked Terrace	1946–1959	1.9	13.4
H15	Post-War Line-built Walk-up Flats	1946–1959	1.8	11.6

Building simulation

Models of the building archetypes were developed in EnergyPlus, a dynamic thermal simulation tool. EnergyPlus version 8 can model airflow through buildings using the validated AirflowNetwork tool and air pollution transport using the Generic Contaminant transport algorithm. The advantage of using a coupled dynamic thermal and contaminant model is that the effect of occupant window-opening behavior in response to internal temperatures can be addressed rather than using fixed schedules. The EnergyPlus

Generic Contaminant model has undergone intermodel comparison against the CONTAM model, with good results (Taylor et al., 2013).

Indoor air simulations were run for the whole year using a Prometheus Test Reference Year (TRY) hourly weather file for Islington, Central London (Eames et al., 2011), with an outdoor $PM_{2.5}$ concentration of 14.7 $\mu g/m^3$ based on the 2010 average background concentration for the GLA (London Air, 2014). Simulations were run with four different orientations of the building (North, West, South, and East). Flats were modeled as being on a middle floor, with adjoining flats to the sides, above, and below. Dwellings with adjoining dwellings to the sides (terraced dwellings, semi-detached, and flats) were assumed to have a net air and contaminant flow of zero between the dwellings, and party walls were not modeled exposed to wind, sun, or polluted external air. Dwellings with adjoining dwellings to the top and bottom (flats) were modeled with identical dwellings above and below and shafts between levels to account for stack effects. Terraced houses were modeled as being mid-terrace with end terraces considered to be semi-detached. Indoor $PM_{2.5}$ levels were output only for mid-floor flats as these represent the majority of dwellings in purpose-built buildings. Local wind speeds were modeled according to an urban terrain, while the solar and wind exposure effects of neighboring but unattached properties were also taken into account.

The infiltration of air was modeled through cracks in the externally exposed facades (walls, roofs, and ground floors of the buildings) and, when open, windows. An even distribution of permeability was assumed across all surfaces, although the net airflow across party walls was assumed to be insignificant at normal operating pressures. Cracks were modeled at the top and bottom of external walls to account for differences in wind pressure according to the height of the building. Vented cellars and lofts were placed above and below the buildings, allowing free movement of outdoor air into these spaces. Cracks in the cellar ceilings and loft floors allowed air from the cellar and loft spaces, respectively, to enter the building based on the defined permeability of the envelope. In the case of flats, air from the cellar and loft entered the ground floor and top floor flats, respectively, and did not directly enter the studied mid-floor flat through these pathways. Internal walls, floors, and ceilings were also given cracks, allowing for the completion of the airflow network and the modeling of stack effects. Cracks were assigned reference air mass flow coefficients based on the building permeability and the surface area, and air mass flow exponents were set to 0.66, as per Jones et al. (2013). Windows and doors were modeled assuming two-way flow.

There are a number of studies that estimate indoor $PM_{2.5}$ deposition and penetration into the building envelope. $PM_{2.5}$ deposition was modeled using a deposition rate of 0.19/h (Long et al., 2001), with a penetration factor of 0.8 when windows were closed and 1.0 when windows were open. These values were used to perform an initial estimation of I/O ratios using a single-compartment box model (Long et al., 2001), typical air change rates of UK dwellings (BRE, 2009), and existing empirical studies of infiltration rates in the UK (Hoek et al., 2008), giving confidence that the values were suitable for modeling UK dwellings. Penetration factor and deposition rate of $PM_{2.5}$ are also highly dependent on particle size (Long et al., 2001); however to simplify analysis, it was modeled as a single contaminant. Indoor pollutant levels and infiltration air change rates (ACH) were calculated every minute and output hourly.

Simulation of typical London dwellings. Two different scenarios were simulated to examine building performance and the influence of occupant behavior:

Scenario 1: No occupant interaction with ventilation components was modeled, and infiltration was only due to the permeability of the externally exposed façades of the dwellings. The dwellings were heated to a set-point of 20°C, and internal gains due to occupant metabolism, hot water, and electrical equipment modeled as per Mavrogianni et al. (2012). Internal doors were assumed to be open at all times, with the exception of bedroom doors, which were closed at night. This represents the base-case performance of the building in terms of pollutant ingress.

Scenario 2: Temperature-driven window opening by building occupants. There are a number of both static and adaptive standards that can be used to estimate the temperature-related comfort of building occupants (CIBSE, 2013). The CIBSE Guide A summertime thermal comfort standards define an upper temperature threshold for comfort of 25°C for living rooms and 23°C for bedrooms (CIBSE, 2006). Internal temperatures were calculated inside the dwelling throughout the year. When internal operative temperatures exceeded the CIBSE summertime thermal comfort standards for living rooms during the day (07:00–22:00) or bedrooms during the night (22:00–07:00), windows were opened in the room. When internal temperatures dropped below the thresholds, they were closed. In both cases, the windows remained closed if the external temperature was greater than the internal temperature. Indoor heating and door-opening behavior was modeled as per Scenario 1. While there is a great deal of uncertainty when modeling building occupant window-opening behavior, the window opening assumptions used are broadly in line with existing field studies of occupant behavior (Dubrul, 1988; Fabi et al., 2012).

Data collation and analysis. Analysis of the hourly indoor $PM_{2.5}$ pollutant predictions of the EnergyPlus models was carried out in SAS 9.3 (SAS Institute, 2013). Occupant exposure to indoor $PM_{2.5}$ was considered to be dependent on the hourly room occupation schedule described in Shrubsole et al. (2012). A script was written to import the EnergyPlus output files and retrieve hourly $PM_{2.5}$ levels from the room occupied at that point in time. The script then calculated the hourly I/O ratio, and then the monthly, hourly–monthly (the ratio for each time of the day, averaged across the month), seasonal, and yearly mean I/O ratio for the simulation period. The results for the typical dwellings were summarized according to the archetype and the occupation scenario. In addition, yearly average ACH values were calculated for the occupied rooms.

London experiences a significant diurnal and seasonal variation in outdoor $PM_{2.5}$ levels. To account for this, the mean hourly–monthly outdoor $PM_{2.5}$ level was obtained (London Air, 2014), and the percent deviation of the temporal values from the background mean calculated. These values were matched against the calculated mean hourly–monthly I/O ratios and used to calculate a temporally scaled I/O ratio for each month and season of the simulation period.

Sensitivity analysis. To explore the sensitivity of the model to variations in input parameters, a differential sensitivity analysis (DSA) was performed for penetration factor, deposition rate, building permeability, retrofit level, wind exposure, London climate, and occupant window and door-opening behavior. The methodology and results of the DSA are discussed further in the Appendix S1.

GIS analysis

Geographical information systems (GIS) data were used to map the spatial variation in the I/O ratio of $PM_{2.5}$ pollution based on the EnergyPlus results and to calculate the absolute indoor concentrations due to outdoor sources only based on predicted outdoor pollution levels. GIS analysis was performed in ArcGIS 10.1 (ESRI, 2013). Data obtained for the research area included the following:

- The GG Building Class topographic map, showing building footprints and building data, such as age and structure type (GG, 2013).
- Ordnance Survey (OS) Address Point data (OS, 2013), showing the number of domestic addresses within each building footprint.
- Department for Environment Food and Rural Affairs (DEFRA) map of estimated outdoor annual mean $PM_{2.5}$ levels across London for 2010 (DEFRA, 2011) (Figure 3).
- Postcode and borough boundary information from the UK Census (UK Data Service, 2013).

The GG Building Class database contains building footprint, built form, and age data for the Greater London Authority. Dwellings were filtered from the Building Class data to remove all non-domestic properties from the analysis. The Building Class database

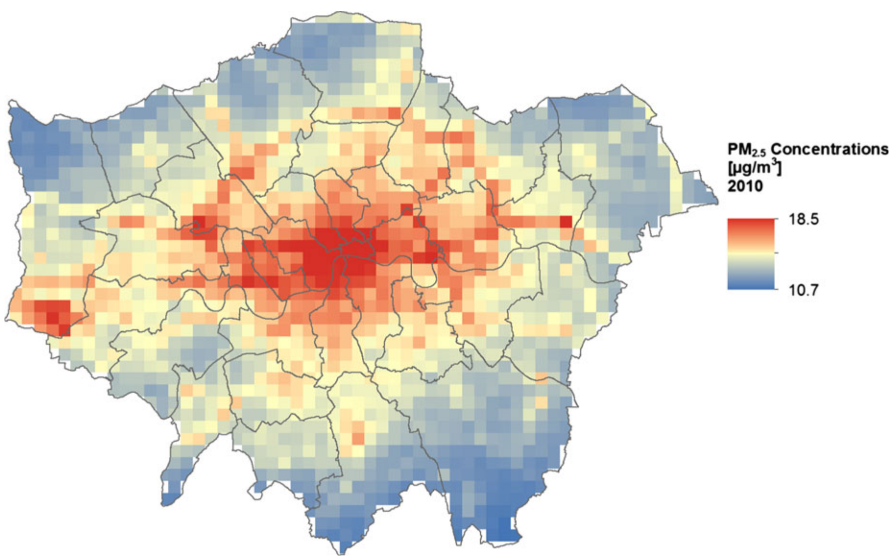

Fig. 3 Estimated outdoor $PM_{2.5}$ concentrations in Greater London for 2010 (DEFRA, 2011)

was filtered further to remove all dwellings that did not have built form or age information, or that did not match the archetypes used in this study. The remaining dwellings accounted for 76% of the known London domestic building stock (and 46% of the total domestic stock), or around 1.5 million dwellings.

The OS Address Point layer was used to determine the number of dwellings within a GG Building Class building footprint, identifying buildings with multiple occupancy. The Address Point layer was filtered to show only domestic addresses within the filtered Building Class footprints. The building classification data from the Building Class database were joined to the domestic address points through a spatial join.

The modeled I/O ratios for the different building archetypes for each scenario were joined to the Address Point database based on the building archetype classifications. The mean I/O ratios of the address points were calculated for each postcode area using the Spatial Overlay tool, and the results mapped to show differences in I/O ratio for dwellings in postcodes across London.

The I/O ratios of dwellings were then used to scale estimated outdoor concentrations of $PM_{2.5}$ to predict absolute indoor concentrations. The DEFRA map for total annual mean outdoor $PM_{2.5}$ concentrations from all sources in the GLA was joined spatially to the address point data. The local outdoor $PM_{2.5}$ concentrations were then used to estimate average monthly indoor concentrations due to outdoor sources only for scenarios 1 and 2, based on the temporally scaled I/O concentration ratios for each month and season, with the assumption that the monthly variation in background $PM_{2.5}$ levels was spatially consistent. The absolute indoor concentrations were summarized by calculating the mean monthly, seasonal, and yearly indoor concentration in each postcode.

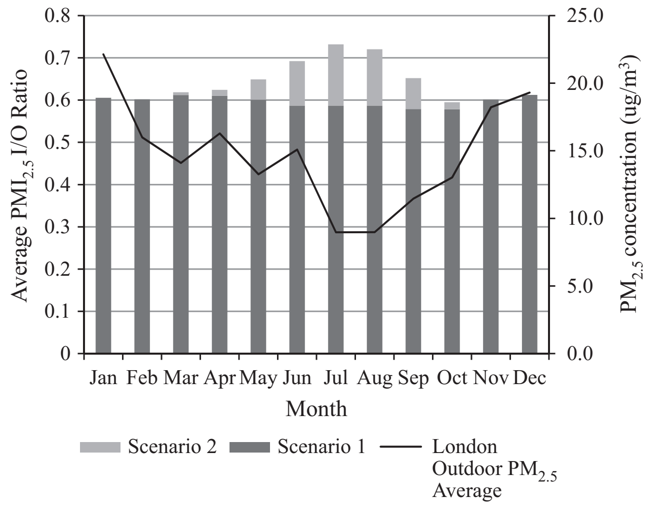

Fig. 4 Average monthly $PM_{2.5}$ I/O ratios for the bungalow living room under scenarios 1 and 2 (modeled with trickle vents), and monthly variation in London background $PM_{2.5}$ levels (London Air, 2014)

Results

Building simulation

An example of the monthly average I/O ratio for a bungalow with trickle vents can be seen in Figure 4. The simulation results showed a slight decline in the $PM_{2.5}$ I/O ratio in Scenario 1 during the summer period (May 1st–August 30th) relative to the winter period (Dec 1st–March 30th), largely attributable to a drop in infiltration caused by a 18% decrease in average wind speeds over this period. Compared with Scenario 1, Scenario 2 predicted an increase in average monthly I/O ratio during the summer period when windows were operable, exceeding winter levels. Sharp short-term increases in the indoor pollutant concentrations could be seen under Scenario 2, with window opening allowing the indoor $PM_{2.5}$ levels to approach the simulated ambient outdoor levels when internal temperatures exceeded the 25°C threshold. Trickle vents were observed to increase the I/O ratios in all buildings.

The EnergyPlus results show a range of annual average I/O ratios of $PM_{2.5}$ concentrations resulting from external sources in dwellings across London (Figure 5). Detached and semi-detached properties with larger permeabilities showed higher amounts of pollution infiltration into the indoor air, while flats showed a much lower I/O ratio of pollution. Opening windows when temperatures exceed a comfort threshold led to an increase in the I/O ratio in all building archetypes, particularly in archetypes prone to overheating during the summer such as purpose-built flats (Mavrogianni et al., 2012). The yearly average ACH for the archetypes can be seen in Table S2. The results of the sensitivity analysis (Appendix S1) indicate that Scenario 1 I/O ratios are highly sensitive to permeability, penetration factor, and deposition rate, and less sensitive to weather file, retrofit level, and occupant window and door-opening behavior. Relative to Scenario 1, Scenario 2 results were more sensitive to retrofit level and less sensitive to permeability and penetration factor, reflecting the influence of temperature-coupled window opening. The degree of parameter sensitivity also varied between archetypes according to the number of exposed external walls, the tendency of buildings to overheat, and the cross-ventilation potential of the dwellings.

GIS analysis

The results of the GIS analysis indicate that many of the dwellings with a higher I/O $PM_{2.5}$ ratio exist outside of Central London (Figure 6). This is likely due to flats being the dominant dwelling type in the more densely populated center, while detached and semi-detached

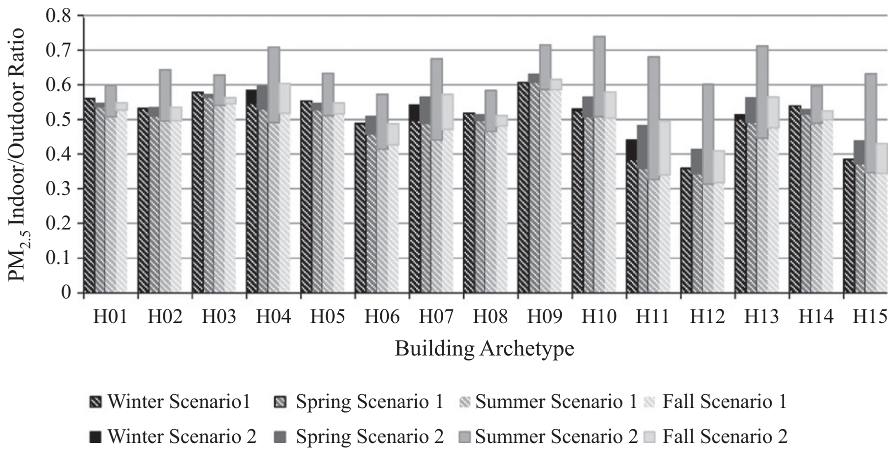

Fig. 5 The seasonal average I/O ratio of $PM_{2.5}$ pollution from outdoor sources, weighted according to room occupancy schedule and estimated frequency of trickle vents

properties are more commonly found in the outskirts of the city. Interestingly, Figure 6 contrasts with many outdoor pollution maps (for example, Figure 3), which show elevated $PM_{2.5}$ concentrations in Central London. There is insufficient building stock data to calculate average I/O ratios for 9.8% of postcodes in the research area. The majority of postcodes with insufficient data are located in Central London, where there are low numbers of residential properties. For Scenario 2, window opening during summer reduced much of the spatial variation seen in other seasons.

The results of the estimated indoor $PM_{2.5}$ concentrations scaled for the DEFRA estimated levels of outdoor pollution can be seen in Figure 7 (Scenario 1) and Figure 8 (Scenario 2). Accounting for the modifying effect of buildings leads to an apparent inversion of the risk of $PM_{2.5}$ exposure when compared to estimates of exposure based on outdoor concentration estimates. Locations with detached and semi-detached dwellings close to pollution sources, such as motorways, major roads, and mainline train tracks, become apparent as having high indoor $PM_{2.5}$ levels from outdoor sources. Maps showing the seasonal variation estimated absolute levels can be seen in the Supplementary Materials (Figure S1). These results indicate that despite an increase in the infiltration due to window opening in Scenario 2 during the summer, the lower outdoor $PM_{2.5}$ levels mean that the absolute indoor concentrations are still higher during the winter.

Discussion

This work has shown how building simulation can be used to determine the indoor $PM_{2.5}$ concentration from outdoor sources in a set of building archetypes, and the results mapped to estimate population exposure in indoor domestic environments. The differences in the $PM_{2.5}$ I/O ratios predicted by EnergyPlus show, in some cases, a two-fold difference between dwelling types, indicating the importance of considering the potentially modifying effect of the building envelope when examining population exposure to air pollution. Occupant behavior can also have a major influence on exposure to outdoor pollution, with simulation results indicating that window opening during hot weather can cause spikes in indoor levels due to outdoor sources of $PM_{2.5}$. Higher infiltration during the summer due to window opening is consistent with existing empirical studies (Hanninen et al., 2011). The results reflect the fact that dwellings with a higher exposed external surface area to internal volume ratio may be more susceptible to higher indoor concentration levels from outdoor sources.

The mapped results of the I/O ratios across London indicate that areas in outer London have higher numbers of detached and semi-detached dwellings that are more susceptible to outdoor pollution infiltration due to their greater externally exposed surface-area-to-volume ratio. This is in contrast to outdoor pollution data, which suggest that higher pollution levels can be generally found in Central London and near major roads and motorways. When $PM_{2.5}$ I/O ratios are scaled against outdoor levels, there is an apparent inversion of exposure risk. The densely populated areas of Central London have the lowest estimated levels of indoor $PM_{2.5}$ from outdoor sources despite the high outdoor concentrations due to attenuation by the predominant built form (flats and terraced dwellings) and their lower fabric permeability. The worst-affected areas were those around a busy circular road (North Circular) and along a major railway routes and highways heading East and West. This study has focused on London; however, the results may provide insight into other urban areas with dense modern flats in the city center and older detached properties in the suburbs or besides major traffic routes.

While there is a lack of empirical data for $PM_{2.5}$ I/O ratios or infiltration rate in UK dwellings, the results

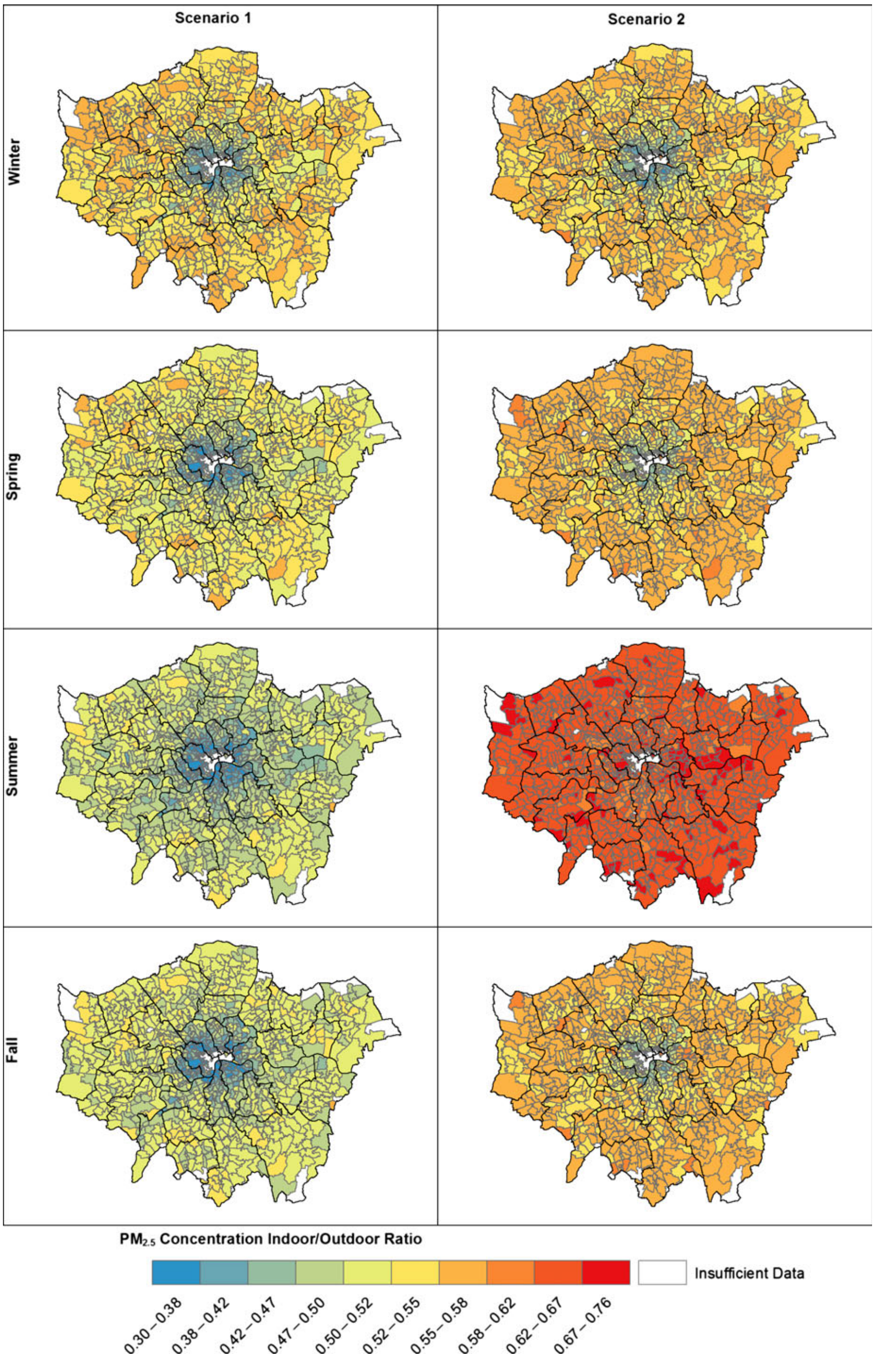

Fig. 6 Seasonal average I/O $PM_{2.5}$ ratios for dwellings across London for Scenario 1 and Scenario 2

are consistent with previous research. The measurements of $PM_{2.5}$ in dwellings in Birmingham estimated an infiltration factor of 0.37 (Hoek et al., 2008), within the range of values obtained in the modeling work detailed. Indoor $PM_{2.5}$ measurements obtained by Nasir and Colbeck (2013) are similar in magnitude, but are difficult to compare directly with modeled results without a schedule of indoor activities, an understanding of the outdoor levels during the measuring period, and building geometry and construction information. Other UK studies have found I/O ratios close to or greater than one due to the presence of indoor sources (Jones et al., 2000; Lai et al., 2006). International studies have found infiltration factors ranging from 0.30 to 0.70 in European studies (Hanninen et al., 2011), and 0.30 to 0.82 internationally (Chen and Zhao, 2011); these values are similar to the 0.33 to 0.60 (Scenario 1) and 0.45 to 0.62 (Scenario 2) ranges predicted by our model. The ACH of the archetypes (Tables S2) are similar to those in empirical studies of UK dwellings

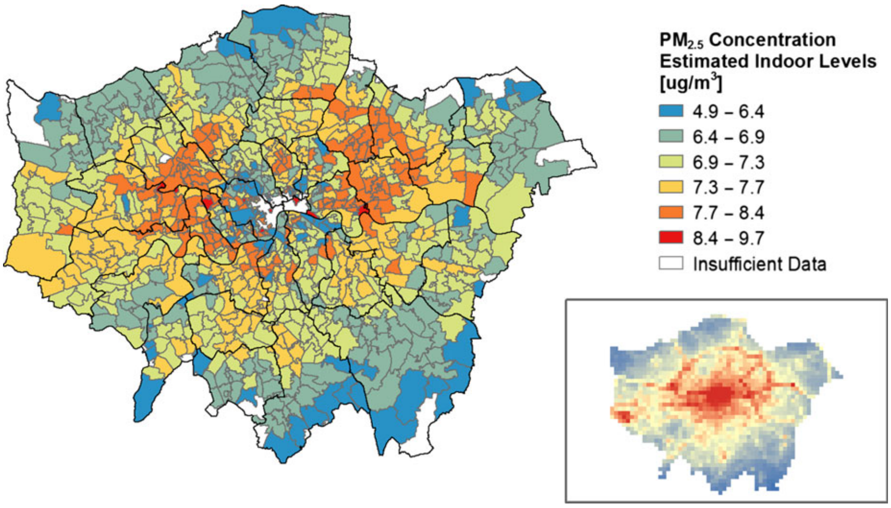

Fig. 7 Estimated absolute indoor $PM_{2.5}$ concentrations from outdoor sources, based on I/O ratio (Scenario 1) and estimated temporal and spatial variations in outdoor concentrations. The inset shows outdoor concentrations from Figure 3

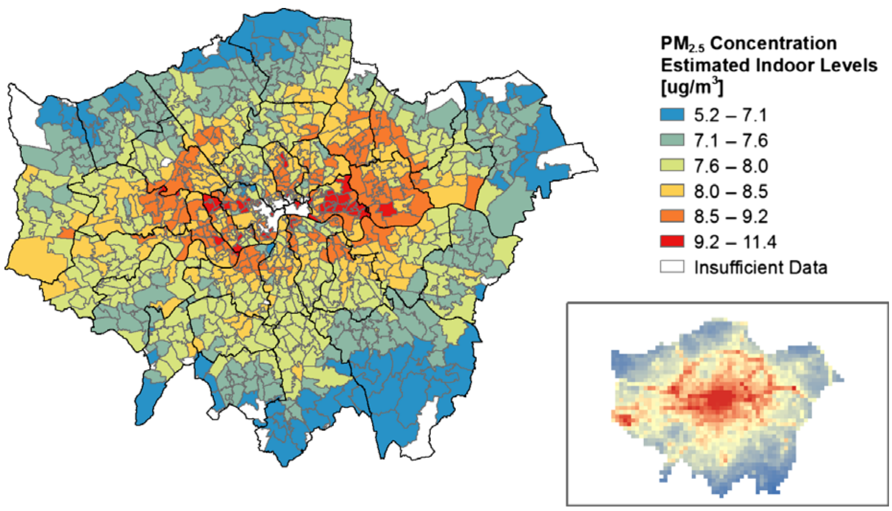

Fig. 8 Estimated absolute indoor $PM_{2.5}$ concentrations from outdoor sources, based on I/O ratio (Scenario 2) and estimated temporal and spatial variations in outdoor concentrations. The inset shows outdoor concentrations from Figure 3

(AIVC, 1994; BRE, 2009; Dimitroulopoulou et al., 2005; Warren and Webb, 1980), while the lower ACH calculated for flats and attached dwellings relative to detached properties have been found in a number of previous studies (e.g. Persily et al., 2006).

There are a number of limitations that need to be considered in this work. While extensive, the coverage of the Building Class database lacked built form and/or age information for 32% of the dwellings in London, and not all of the known dwellings had a relevant archetype, which was modeled. Developing building archetypes for each combination of built form and age is unrealistic and would take a significant amount of time to simulate using currently available building simulation tools. The archetypes are intended to represent average buildings in London rather than a specific property, and deviations of individual buildings from the nominal archetypes are minimized when the results are considered over a wider geographical scale – in this case, postcode. Mid-floor flats are assumed to represent the majority dwelling type in multidwelling buildings and represent an 'average' of potential stack effects. It was not possible to model flats at different levels, as there is no information on the vertical distribution of addresses in the London building stock. The $PM_{2.5}$ levels outside dwellings toward the top of the building are likely to be lower than levels at the bottom due to the generally larger distance from outdoor sources and the influence of local meteorology effects, specifically increases in wind speeds (Vardoulakis et al., 2008); however, higher wind pressures may increase infiltration rates. All simulations were run with wind speeds modified to reflect an urban terrain; however, terrain type in London varies from densely built central city areas to less dense outer suburbs. The increased exposure to wind forces in suburban areas is expected to lead to higher I/O ratios and potentially an increase in the apparent inversion of risk.

Internal sources are an important contributor to indoor $PM_{2.5}$ levels, but have not been included in this study. While some building types allow higher levels of outdoor $PM_{2.5}$ infiltration due to a high ACH, such buildings may also have a greater ability to allow indoor-produced $PM_{2.5}$ to exfiltrate. This may mean that occupants of different building types may be exposed to different ratios of indoor-sourced to outdoor-sourced $PM_{2.5}$. The chemical and toxicological profile of indoor sources of $PM_{2.5}$ may differ from that of outdoor sources, meaning that they may lead to different health effects (Wilson et al., 2000).

Assumptions were also required in modeling the occupant behavior in Scenario 2. Window-opening behavior is complex, and indoor temperature is not the sole driver. Furthermore, top-level flats are more susceptible to overheating (Mavrogianni et al., 2012), a fact which suggests that occupants may open windows more frequently to reduce the internal temperatures and therefore temporarily drive up indoor levels of outdoor pollutants.

Only domestic properties were modeled and mapped in this study. While research suggests that people in the London spend over 80% of their time indoors (Kornartit et al., 2010), this includes time spent at work in, for example, offices, or engaging in leisure activities in shopping malls and theaters. Nonetheless, epidemiological analyses typically use the home postcode as an indicator of exposure, and this research is able to offer insight into how their dwellings may influence this exposure. This study has examined the indoor pollution levels throughout the day as an indicator of building performance and as such does not consider the fact that certain socio-demographic groups may spend a longer time than others in their home.

The modeling methodology used also carries with it a number of uncertainties. The EnergyPlus airflow network model is based on a validated airflow model, and initial comparisons between it and the indoor air quality model CONTAM give confidence in the results for contaminant transport (Taylor et al., 2013). Air leakage paths were assumed to be distributed across all bounding surfaces in the dwellings including party walls, which were assumed to be fully permeable. In reality, party walls may contribute up to 30% of air leakage at 50 Pa pressure differential in UK dwellings (Stephen, 2000). The calculated distribution of permeabilities for the EHS dwellings matched the measured distribution from the research of Stephen (2000) when the sheltering factor was included in calculations and was slightly higher when sheltering was excluded. Using the slightly higher values for buildings with party surfaces (equivalent to 22.5% higher for flats with three bounding surfaces) and applying them only to external walls, an attempt was made to compensate for the differences in permeability between external and party walls. However, further research is required to understand the permeability of different buildings and surface types in the UK housing stock. Modeling $PM_{2.5}$ as a single contaminant is an important simplification and must be acknowledged.

Retrofit and airtightness measures, such as draught proofing, replacement windows, loft insulation, and the sealing of suspended floors, can reduce the permeability of a dwelling (Hong et al., 2004). There has been a significant focus on decarbonizing dwellings in the UK by limiting the heat loss through the building envelope. Building regulations specifying the air tightness requirements for new dwellings, as well as retrofits to reduce the permeability of existing structures, are one of the means to achieve energy use reductions. These measures will have the additional benefit of reducing pollutant infiltration into dwellings and reducing the I/O ratios of outdoor pollutants.

While a number of assumptions were necessary for this research, the results provide an insight into the potential modifying effects of the built form and building envelope on pollution infiltration in the London dwelling stock. Further field work is required to confirm the influence of built form and building permeability on the infiltration of outdoor pollution indoors. This research has implications for assessing the population exposure to pollutants from outdoor sources and can be used to supplement existing research into indoor air quality in London. Future research will increase the number of building archetypes to be representative of the entire UK, while additional pollutants will also be modeled from both outdoor and indoor sources.

Conclusions

This analysis has mapped the potential indoor exposure of the London population to different $PM_{2.5}$ levels from outdoor sources based on domestic building stock characteristics. The relative vulnerability of different dwellings to $PM_{2.5}$ ingress has been demonstrated, and dwelling stock databases used to indicate areas where the stock is most vulnerable to high outdoor pollutant levels. This research indicates that flats have a reduced I/O ratio for $PM_{2.5}$ from outdoor sources when compared to detached and semi-detached dwellings. The higher concentration of flats in Central London leads to an apparent inversion of exposure to indoor $PM_{2.5}$ from outdoor sources when compared to estimates of exposure based on outdoor concentration estimates. The results can provide insight into other urban areas with spatial variations in building stock and outdoor pollution levels.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Supporting information.
Table S1. Thermal conductivity properties of the building envelope.
Table S2. Modeled annual average Air Change Rates (h-1) or (ACH) for the dwelling archetypes under the two scenarios.
Table S3. Variations in parameter inputs for DSA.
Table S4. Variations in yearly average I/O ratio for Scenario 1 for different parameters.
Table S5. Variations in summer average I/O ratio for Scenario 2 for different parameters.
Figure S1. Seasonal changes in estimates absolute indoor and outdoor $PM_{2.5}$ levels across London.

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A tale of two cities: Comparison of impacts on CO_2 emissions, the indoor environment and health of home energy efficiency strategies in London and Milton Keynes

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HIGHLIGHTS

- We model current the (2010) and future (2050) housing of London and Milton Keynes.
- We apply a range of building interventions and grid decarbonisation scenarios.
- We investigate impacts on energy use, indoor temperatures, pollutants and health.
- Results vary appreciably both between locations and the direction of health impacts.
- Locationally tailored policy approaches are needed to achieve CO_2 reduction targets.

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ABSTRACT

Dwellings are a substantial source of global CO_2 emissions. The energy used in homes for heating, cooking and running electrical appliances is responsible for a quarter of current total UK emissions and is a key target of government policies for greenhouse gas abatement. Policymakers need to understand the potential impact that such decarbonization policies have on the indoor environment and health for a full assessment of costs and benefits. We investigated these impacts in two contrasting settings of the UK: London, a predominantly older city and Milton Keynes, a growing new town. We employed SCRIBE, a building physics-based health impact model of the UK housing stock linked to the English Housing Survey, to examine changes, 2010–2050, in end-use energy demand, CO_2 emissions, winter indoor temperatures, airborne pollutant concentrations and associated health impacts. For each location we modelled the existing (2010) housing stock and three future scenarios with different levels of energy efficiency interventions combined with either a business-as-usual, or accelerated decarbonization of the electricity grid approach. The potential for CO_2 savings was appreciably greater in London than Milton Keynes except when substantial decarbonization of the electricity grid was assumed, largely because of the lower level of current energy efficiency in London and differences in the type and form of the housing stock. The average net impact on health per thousand population was greater in magnitude under all scenarios in London compared to Milton Keynes and more beneficial when it was assumed that purpose-provided ventilation (PPV) would be part of energy efficiency interventions, but more detrimental when interventions were assumed *not* to include PPV. These findings illustrate the importance of considering ventilation measures for health protection and the potential variation in the impact of home energy

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efficiency strategies, suggesting the need for tailored policy approaches in different locations, rather than adopting a universally rolled out strategy.

1. Introduction

Housing is responsible for one quarter of the UK's total end-user CO_2 emissions, half of which comes from space heating (Hamilton et al., 2009; DECC, 2011). Motivated by CO_2 emissions reduction targets, fuel poverty, energy security and in response to the EU Energy Performance of Buildings Directive (EPBD), the UK Government is implementing policies designed to make major improvements to the energy performance of the housing stock (DECC, 2009; EU, 2011a). In order to meet the UK's ambitious target of an 80% reduction in CO_2 emissions from 1990 levels by 2050 (DECC, 2012a), a number of programs and policies are being employed that aim to increase dwelling airtightness and increase fabric performance through the provision of insulation, glazing upgrades and improvements in heating systems. The UK Government estimates that fabric efficiency measures including cavity, solid wall and loft insulation could result in energy savings of ~12 TWh by 2020 (DECC, 2014). As existing dwellings are predicted to represent 70–80% of the 2050 building stock (Palmer and Cooper, 2011), much of the energy efficiency gains must be obtained through retrofitting of the existing stock. This requires substantial investment, as nearly all of the UK's 26.4 million dwellings will require an upgrade in energy performance to meet emission reduction targets (CCC, 2010; EU, 2011b). Interventions applied to reduce ventilation heat loss such as draught stripping and double glazing impact the airtightness of dwellings (Hong et al., 2004). Making dwellings more airtight without additional purpose provided ventilation (PPV) increases the risk of exposure to higher concentrations of indoor sources of pollutants such as $PM_{2.5}$, mould, environmental tobacco smoke (ETS) and radon, whilst reducing ingress of externally sourced contaminants and increasing indoor winter temperatures (Bone et al., 2010; Milner et al., 2014). However, increasing PPV to improve indoor air quality(IAQ) may result in a reduction in energy efficiency gains through ventilation heat loss (Godish and Spengler, 2004), which on average in the UK is estimated to account for 12% of a dwellings total energy use (Hamilton et al., 2009). The trade-off between these different policy objectives (energy conservation and ventilation for health) has been previously noted (Crump, 2011). With the UK population spending on average around 80% of their time indoors, and around 50% of their time in their homes (Kornartit et al., 2010), building are important modifiers of population health (Thomson et al., 2013). Recent UK based monitoring has shown that homes with higher energy performance levels are associated with a higher risk of diagnosed asthma (Sharpe and Shearer, 2014), suggesting that retrofits as currently implemented can have negative effects on household health. In addition, poorly designed interventions could lead to a range of unintended consequences across multiple domains (Shrubsole et al., 2014). Yet, if measures are properly designed, applied and operated, it is probable that they could have major net benefits for public health (Wilkinson et al., 2009).

National targets for CO_2 emissions reduction in the UK, one of the main drivers of changes in energy performance in dwellings (Rosenow, 2012), are set out in the Climate Change Act of 2008 (HM Government, 2008). Individual sectors such as housing, have contributory targets (CCC, 2010). Total emission reductions from the housing stock will occur through energy efficiency interventions and by decarbonizing the dwelling energy supply. The carbon intensity (CI) of the supply grid will influence future CO_2 emissions depending on the mix of sources, e.g. coal or gas fired power stations, renewables and nuclear. Scenarios have been described for decarbonizing the grid, in line with emissions reductions targets (CCC, 2010). These changes to power generation, in conjunction with policies aimed at transport and industry are also expected to reduce airborne pollution, improving future air quality (Williams, 2007). The coupling of fuel source and power grid decarbonization scenarios with energy efficiency retrofits to the housing stock and the impact on IAQ and health is a relatively new area of research, although they are increasingly recognized as factors in achieving UK wide CO_2 reduction targets by Government (DECC, 2013).

In this paper we describe a modelling study to assess changes in energy use, CO_2 emissions, winter indoor temperatures, indoor airborne pollutant concentrations and associated impact on health of selected combined home energy efficiency and electricity grid decarbonization scenarios. We apply these scenarios to London and Milton Keynes. London, a major city with an estimated 2010 population of 7.83 million, responsible for 8.4% (44.71 Mt) (GLA, 2010) of UK CO_2 emissions, and characterized by both new and old buildings and higher density forms; Milton Keynes, 72 km northwest of London, created under the UK's 2nd New Towns Act 1965, with an estimated 2010 population of 241,500, and responsible for 0.3% (1.76 Mt) of UK CO_2 emissions (MKiO, 2014), with predominantly newer and low-density forms. Population figures for 2010 are used to coincide with scenario start dates and inform health calculations.

2. Methods

2.1. Modelled scenarios: decarbonization of the housing stock and electrical grid

We modelled the current stock (2010) and the impact of three future housing/electricity grid decarbonization scenarios applied to the housing stock in London and Milton Keynes (Table 1).

Starting from the 2010 stock in both locations, interventions were applied that brought the 2050 stocks to parity in order to quantify the possible health impacts, energy use and CO_2 savings. The future contrasting scenarios are:

(i) 'Energy Efficient (EE)': This assumes a business-as-usual trajectory with regard to the carbon intensity of the electricity grid to 2050 with a range of housing interventions applied to all properties not currently having them. For housing interventions, data on the existing measures in the current stock (2010) were derived from a variety of empirical sources (EHS, 2012; CSE, 2012; HECA, 2013; HEED, 2014; MKiO, 2014).

(ii) 'Energy Efficiency Plus (EE+)': This assumes business-as-usual carbon intensity of the grid, but with additional housing interventions focused on heating and seeks to investigate the impact of a greater focus on technical adaptation of dwellings. These are applied to all properties

Table 1
Combined future scenarios for grid decarbonisation and housing energy interventions.

Grid decarbonisation scenario	Range of energy efficiency and ventilation interventions[a]	Source	CI[b] 2010	CI 2020	CI 2030	CI 2050
Energy efficiency: A range of energy efficiency and ventilation housing interventions with no decarbonisation of the electricity grid.	Draught stripping New double glazing with trickle vents Extract fans Cavity wall filling Solid wall insulation Insulate lofts to 250 mm[c]	UKERC	464	480	420	360
Energy Efficiency Plus: Substantial energy efficiency and ventilation housing interventions occur with no decarbonisation of the grid.	Draught Stripping New double glazing with trickle vents Extract fans Cavity wall filling Solid wall insulation Insulate lofts to 250 mm Install condensing boilers Central heating	UKERC	464	480	420	360
Low Carbon Supply: An ambitious scenario; a range of energy efficiency and ventilation housing interventions occur at the early stages to reach UK interim targets. Major decarbonisation of the electricity grid.	Draught stripping New double glazing with trickle vents Extract fans Cavity wall filling Solid wall Insulation insulate lofts to 250 mm	UKERC	464	290	70	25

Arrow denotes direction of increasingly aggressive supply decarbonisation scenarios.
[a] Energy efficiency and ventilation interventions are applied to those houses not having them according to the English Housing Survey (EHS 2010).
[b] CI (carbon intensities) expressed in grams of CO_2 per kWh. These are estimates of equivalent CO_2 emissions normalized per unit of delivered electricity (i.e. including transmission and distribution losses).
[c] Loft insulation topped up to or installed to 250 mm (BRE, 2009).

currently without them and therefore represent the upper bound case.

(iii) ‘Low Carbon Supply (LCS)’: This assumes an aggressive supply decarbonization scenario with housing interventions as in (i) and that space heating in houses will be 100% electrified by 2050.

All scenarios and their individual components start from a baseline of 2010 and were specified to coincide with the CO_2 reduction target date of 2050.

The grid decarbonization scenario used in (i) and (ii) are equivalent to the ‘resilient’ scenario, whilst (iii) is equivalent to the ‘Low-carbon’ scenario both seen in the UKERC Research Report (UKERC, 2013).

The baseline (2010) figure for carbon intensity (CI) comes from data for centralized electricity generation from the Digest of UK Energy Statistics, DUKES (2011). The emission reductions of the energy supply grid and power sector fuel mix and grid emission figures for carbon intensity (CI) were derived from the UK Energy Research Centre (UKERC) scenarios within the UK Committee on Climate Change 4th Carbon Budget report. (CCC, 2010; UKERC, 2013). These were chosen as they allow for structural uncertainties in future energy supply and represent the upper and lower boundaries of possible grid decarbonization. The year by year CI figures represent national targets with local trends assumed to evolve similarly over time. UKERC scenarios reduce grid emissions through specific investment choices, such that remaining sector reductions (including housing) are to be achieved through technology, efficiency and conservation (UKERC, 2013).

We assumed that installed measures are replaced once their life expectancy is over (e.g. boilers are replaced after 15 years). Due to uncertainties, we made no allowance for possible future improvements in efficiency or new technology.

Although the UK building regulations require that air quality is made no worse following retrofitting, there is no specific guidance regarding ventilation for energy efficiency retrofits. All future scenarios were run specifying the inclusion of purpose-provided ventilation (PPV) (extract fans and trickle vents) to maintain adequate air exchange following airtightening in accordance with current Building Regulation requirements for new builds (HM Government, 2010). Given the potential importance that ventilation has on air quality in the home (Bone et al., 2010), additional simulations without PPV were run so as to examine the importance of ventilation characteristics for impacts on CO_2 emissions and health.

2.2. Modelling the scenario impacts: the SCRIBE model

Modelling of scenario impacts was carried out using a UK housing stock computer model known as SCRIBE (Strategies for Carbon Reduction In the Built Environment), developed by University College London (UCL) and the London School of Hygiene and Tropical Medicine (LSHTM) (Hamilton et al., 2015). SCRIBE incorporates (i) a building physics module that enables estimation of energy use, indoor environmental conditions (winter temperatures and annual pollutant concentrations) and mean CO_2 emissions under a range of housing interventions and projected changes in grid carbon intensities, and (ii) a model of health impacts associated with indoor environmental conditions. Modelled mean CO_2 emission reductions are compared to emission levels required to meet targets set in the Climate Change Act 2008, from a baseline of 2010, rather than 1990 (HM Government, 2008). The baseline of 2010 is used due to limitations in data availability for some SCRIBE inputs, particularly building/intervention data for Milton Keynes prior to this date. Consequently, CO_2 emission reductions targets are adjusted as follows: for 2020–43%, for 2030–57% and for 2050–75%, all relative to 2010 instead of 1990. This adjustment has no impact on the 2050 results. Details of the SCRIBE model and the inputs used in the various components are outlined in Fig. 1.

2.3. Modelling indoor air quality

Within the SCRIBE tool outputs are produced using CONTAM, a validated airflow and pollutant transport building physics tool (Emmerich, 2001). Geometries representative of the London and Milton Keynes housing stocks were constructed to assess changes to the indoor environment (air quality, winter temperature and energy use) associated with interventions of dwellings in the 2010 English Housing Survey (EHS, 2012). Ten dwelling geometries are used based on Oikonomou et al. (2012) and Wilkinson et al. (2009), supplemented with typical floor plans and facades available from the literature. The resultant built forms are matched to each EHS entry using criteria of dwelling type and size. Outline plans and model screen shots for all archetypes are shown in the online supplementary file accompanying this study.

For baseline (2010) indoor pollutant concentrations, each geometry in CONTAM is remodelled with four distinct ventilation system options: (i) no trickle vents or extract fans, (ii) trickle vents only, (iii) extract fans only and (iv) trickle vents and extract fans. This gives a total of 40 dwelling form-ventilation archetypes with which to represent the EHS dwelling variants. All ventilation components are assumed to be functioning correctly with no allowance made for mechanical failure or deterioration with time. It is acknowledged that this could lead to slightly lower indoor pollutant concentrations. Each of the 40 archetypes is modelled with eight permeabilities ranging from 3 to 30 $m^3/h/m^2$@50 Pa present in the English stock (Stephen, 1998), giving a total of 320 archetypes. Each EHS variant is mapped to one of these models using the predicted permeability value. These are simulated in CONTAM, to obtain concentrations of indoor and outdoor sourced particulate matter ≤2.5 μm ($PM_{2.5}$), radon, environmental tobacco smoke (ETS), and moisture (as a precursor of mould). Under each scenario the adapted EHS variants are mapped to the CONTAM models to reflect the change in permeability following the interventions, thus future changes in indoor pollutant concentrations are estimated. This mapping includes anticipated reductions in external $PM_{2.5}$ concentrations, specified by year and location. For London, the 2010 annual mean outdoor urban background concentration $PM_{2.5}$ is taken as 13.0 μg m^{-3} (Shrubsole et al., 2012). For Milton Keynes, the figure of 10.9 μg m^{-3} is based on data from the Defra mapping project (Defra, 2013). For future $PM_{2.5}$ concentrations, Defra data is available to 2030 in both locations. For 2050, a linear trend is assumed, in order to bring results in line with 2050 predictions from Williams (2007). The SCRIBE model differentiatesPM$_{2.5}$ from indoor and outdoor sources due to differences in particle nature and potential (but largely unquantified) relative toxicity, which are sufficiently great as to require separate consideration (Rohr and Wyzga, 2012)giving health impact assessments the opportunity to distinguish relative risks to population health.

Radon exposures are informed by the national distribution reported in Gray et al. (2009) and adapted to allow for regional differences in emission rates (HPA, 2011). Due to low levels of radon in London and Milton Keynes - geometric means 16 and 43 Bq/m^3 respectively (HPA, 2011) – we have not proposed any specific radon remediation measures in our modelling. Smoking levels are informed by NHS, 2011. No account is taken of any future changes in smoking prevalence, or outdoor smoking behaviour that may influence exposure for non-smokers in smoking households. Indoor pollution emission profiles are derived from empirical studies (Table 2). Within the CONTAM modelling each pollutant has a defined source and emission period: indoor $PM_{2.5}$ is a function of occupancy and cooking; moisture is a function of occupancy and bathroom use; and ETS is a function of occupancy, with weekend and weekday occupancy profiles differing. Readers are directed to the on-line supplementary data accompanying this study for full details.

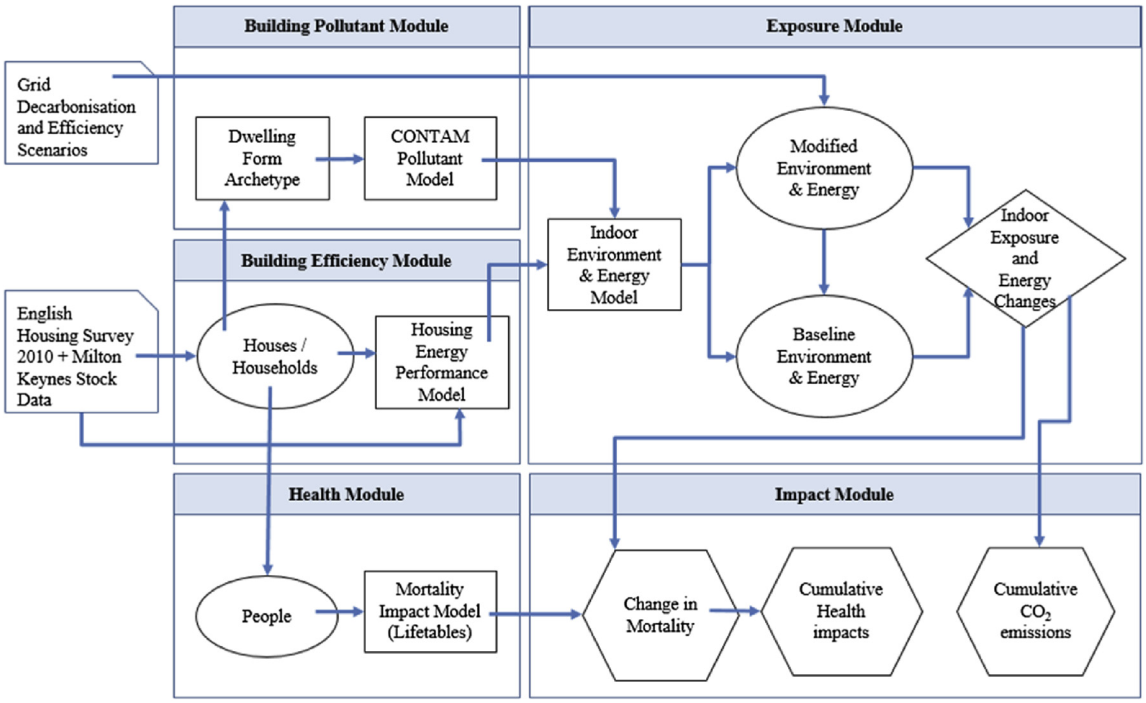

Fig. 1. Connections between grid decarbonisation and energy efficiency and ventilation measures in housing and the impacts on health and CO_2 emissions within the SCRIBE tool (adapted from Hamilton et al., 2015).

Table 2
Data sources for indoor pollutant inputs.

Pollutant	Values	Data source
$PM_{2.5}$ emission	Cooking 1.6 mg/min	Ozkaynak et al. (1996), Chen and Zhao (2011)
$PM_{2.5}$ deposition	0.39 l/h	
Radon	0.005 Bq, 0.05 Bq and0.1Bq (one decay s^{-1}) x's room floor area m^2	Fang and Persily (1995), Gray et al. (2009), HPA (2011)
Environmental Tobacco Smoke	0.99 mg/min at 5 min per cigarette	NHS (2011), He et al. (2004), Afshari et al. (2005)
Moisture (precursor of mould)	Various values depending on source (n = 7)	FMNectar (2007), Gilbertson et al. (2012)

See on-line supplementary document for full details and schedules of emission and activities.

2.4. Modelling changes in indoor air temperature and air quality

For London, existing energy and ventilation interventions are modelled directly from the English Housing Survey (EHS, 2012), which comprises a representative sample of properties (16,150 surveyed dwellings) with weights for each dwelling variant which can be used to represent all households in England. Regional information enables London dwelling variants to be directly selected and used in the modelling. The survey does not have a sufficient or identifiable sample for Milton Keynes.Dwellings were therefore simulated by either using alternative empirical data sets for the variables required for SCRIBE; for example the range of existing interventions and dwelling age and type (CSE, 2012; HECA, 2013; HEED, 2014; MKiO, 2014). For the few remaining variables that were not available: (i) the Standard Assessment Procedure[1] (SAP) rating, (ii) envelope permeability and (iii) ventilation type; the *known* variables were used to calculate estimates for SAP rating and envelope permeability. The probability of occurrence in each of the ~16,000 EHS variants were then used to randomly sample the Milton Keynes housing stock and scaled to the correct number of dwellings.

The stock modelling input variables and their ranges are shown in Table 3.

The building efficiency module estimates envelope permeability and heat loss resulting from fabric performance, heating system and ventilation characteristics. A conversion process uses EHS variables to infer features, such as dwelling geometry and construction characteristics to predict ventilation and thermal performance (DECC, 2012b). The SAP criteria is then used to predict total ventilation rate, dwelling permeability, and fabric heat loss rate (Hughes et al., 2013). These are combined with the heating system performance to predict a heat transfer characteristic E-value[2] for each dwelling, using a relationship that takes into account the expected behaviour of the occupant (Oreszczyn et al., 2006). Each intervention is associated with changes in the thermal and ventilation characteristics of the EHS variants. The SAP method is used to calculate the new heat transfer characteristic, and the new E-value is predicted such that changes in energy use can be estimated under the variety of future scenarios. These include PPV, designed to comply with Approved Document F1 (HM Government, 2010). One of the key assumptions in the health impact modelling (section 2.5) is that additional PPV will be installed in dwellings alongside the energy efficiency measures. For outdoor conditions influencing indoor values, transient yearly weather files are constructed using Chartered Institution of Building Services Engineers (CIBSE) Test Reference Year (TRY) and Design Summer Year (DSY) data. The TRY is a synthesized typical weather year suitable for analysing the environmental performance of buildings, whereas the DSY is a complete historical year representing a near extreme warm summer (CIBSE, 2010). These files contain hourly outdoor air temperature, air pressure, wind speed, wind direction, and humidity data.

2.5. Health impacts

The health impacts associated with changes to annual indoor air quality and heating season temperatures, were modelled within the SCRIBE tool using life table methods based on the IOMLIFET model (Miller and Hurley, 2003) using all-cause and cause-specific mortality data for England and Wales available from the Office for National Statistics (ONS), with separate life tables for males and females. The key model output was changes in years of life lived with no morbidity estimates. Exposure-response relationships for changes in indoor exposures (i.e. standardized internal temperature (SIT), ETS, $PM_{2.5}$ derived from indoor and outdoor sources and radon) were derived from published sources shown in Table 4. Where more than one exposure was related to the same outcome, we assumed that the risks are multiplicative in line with the work of Scarborough et al. (2010).

Health impacts were modelled year by year to 2050. For all modelled outcomes other than those associated with changes in SIT, we specified outcome-specific inception and cessation lag functions to reflect the time delay between changes in exposure and subsequent change in disease status. See Hamilton et al. (2015) for further details.

3. Results

3.1. Energy use and CO_2 emissions

Changes in Energy consumption (kWh) and CO_2 emissions of the housing stocks relative to the 2010 baseline taking account of the changing carbon intensity (CI) of the electrical grid are shown for each scenario in Table 5. Values are expressed as the percentage increase relative to the base year of 2010, with negative figures therefore indicating reduction in CO_2 emissions.

Greater reductions in energy use are seen in London relative to Milton Keynes under all intervention scenarios, with the highest gains seen in the EE + scenario where no additional purpose provided ventilation (PPV) was assumed. For both the EE and EE + scenarios, appreciably greater CO_2 reductions were seen in London than in Milton Keynes. Aggressive decarbonization of the electric grid, combined with housing measures in the LCS scenario exceeded the targets needed for compliance with the Climate Change Act, 2008 in both locations. The addition of ventilation interventions increased energy use by an average of 8.6% across the scenarios.

3.2. Temperature and pollutant concentrations changes

Table 5 shows the changes that occur in mean indoor temperature during the heating season and annual airborne pollutant concentrations following the installation of both energy efficiency

[1] SAP: The Government's Standard Assessment Procedure for Energy Rating of Dwellings (BRE, 2012).

[2] The E-value represents the dwelling heat transfer characteristic, obtained by combining the estimated fabric and ventilation performance with the heating system (after Oreszczyn et al., 2006).

Table 3
Stock modelling input variables and their ranges.

Stock Variables	Variable range	London stock source	Milton Keynes stock source
Dwelling types	End terrace, mid terrace, semi detached, detached, bungalow, converted flat, purpose built flat-low rise, purpose built flat-high rise	English Housing Survey (EHS, 2012)	2011 Census; MKiO, 2014
Dwelling age	Pre 1919, 1919–44, 1945–64,1965–80, 1981–90, post 1990		2011 Census; MKiO, 2014
Wall types	Cavity with insulation[a], cavity uninsulated, solid uninsulated[b]		CSE, 2012
Glazing types	Mixed[c] Single[d] Double		CSE, 2012; MKiO, 2014
Eligible for loft insulation	Yes, No[a]		CSE, 2012; HEED, 2014; MKiO, 2014
Eligible for boiler upgrade	Yes, No		CSE, 2012; MKiO, 2014
Eligible for central heating upgrade	Yes, No		CSE, 2012; HECA, 2013; MKiO, 2014
Ventilation type	No trickle vents or extract fans, trickle vents only, extract fans only trickle vents and extract fans		EHS, 2012,[e] and allocated as under Section 2.4
SAP level	<30, 30–50, 51–70, >70		EHS, 2012,[e] and allocated as under Section 2.4
Permeability@50 Pa	3, 5, 7, 10,15, 20, 25, 30	Distribution: Stephen, 1998, 2000	EHS, 2012,[e] and allocated as under Section 2.4

[a] Likely slightly underestimates totals for Milton Keynes as does not include properties receiving measures prior to 2005 as no reliable data exists (CSE,2012).
[b] Assumes all solid wall properties have the potential for insulation.
[c] Lack of data for mixed types.
[d] Assumed if not double glazed.
[e] As no data available, weighting for South East region used.

Table 4
Modelled mortality outcomes and exposure-response relationships.

Exposure	Health outcome	Exposure-response relationship	Ref.
Standardized internal temperature	Winter excess cardiovascular	0.98 per °C	Derived from Wilkinson et al. (2001)
Environmental tobacco smoke	Cerebrovascular accident	1.25 (if in same dwelling as smoker)	Lee and Forey (2006)
	Myocardial infarction	1.30 (if in same dwelling as smoker)	Law et al. (1997)
$PM_{2.5}$	Cardiopulmonary	1.082 per 10 μg/m^3	Pope et al. (2002, 2004)
	Lung cancer	1.059 per 10 μg/m^3	As above
Radon	Lung cancer	1.16 per 100 Bq/m^3	Darby et al. (2005)

Table 5
Changes in mean *per capita* energy consumption and CO_2 emissions of the London and Milton Keynes housing stock under each of the scenarios with and without purpose-provided ventilation. Negative values signify reduction in energy or CO_2 emissions compared with 2010.

Location	Scenario	% Change in mean energy use 2010–2050		% Change in mean CO_2 emissions 2010–2050	
		With purpose-provided ventilation	Without purpose-provided ventilation interventions	With purpose-provided ventilation	Without purpose-provided ventilation interventions
London	EE	−37.73	−35.51	−50.00	−51.65
	EE+	−44.85	−42.87	−55.73	−57.25
	LSC	−37.73	−35.51	−96.56	−96.69
Milton Keynes	EE	−16.68	−14.86	−33.86	−35.33
	EE+	−26.54	−24.91	−41.65	−43.00
	LCS	−16.68	−14.86	−95.60	−96.05

and PPV interventions under the three scenarios for 2050.

A mould risk of >1 indicates the likely presence of mould in a property, with figures representing the % of properties where the mould risk is > 1. A decrease shows a reduction in health risk. However, these changes in mould risk have not been used in the calculation of health impact for this work (though there is some evidence of likely impact, particularly in children). For scenarios with PPV Higher mean indoor temperatures during the heating season are seen in the housing stock following retrofitting, with appreciable reductions in most of the pollutants studied except for radon gas, which shows small reductions in both locations. As a continuous source radon is not appreciably dissipated by intermittent ventilation measures such as extract fans. However, the values seen are typically low for these cities and well below the 200 Bq/m^3 action level (HPA, 2011). In Milton Keynes, the housing stock is more recent and therefore built to a higher energy efficiency standard and greater airtightness. The housing typology also differs appreciably with over 50% of London's stock being purpose built flats (requiring simpler measures to obtain gains), while Milton Keynes stock comprises 80% detached, semi-detached and terraced dwellings with general larger building volumes (MKiO, 2014). This is reflected in the greater reduction in indoor sourced pollutants seem in the London stock.

The changes in mean indoor heating season temperatures were only marginally greater without PPV than in the scenarios which included it. The ingress of $PM_{2.5}$ derived from the outdoor air was reduced by the greater airtightness without PPV, but concentrations of all other pollutants showed increases from the 2010 baseline for both locations. London dwellings suffer more, due to greater relative reduction in envelope permeability and therefore air-tightness. The exception is radon because of the greater emission levels seen in Milton Keynes (which are determined by local geology) (HPA, 2011).

3.3. Health impacts

The impact on health measured in terms of the *per capita* total of life years gained (Table 7) is greater in magnitude in London than in Milton Keynes. These impacts translate into increases in average life expectancy at birth of ~3 months (Milton Keynes) and ~4 months (London) with PPV, but *decreases* in life expectancy of ~2 months (Milton Keynes) and ~5 months (London) if PPV is not installed. This reflects the larger changes in the modelled indoor exposures in London, which are due to the greater potential for improving the housing stock primarily due to the greater age range of the London stock (generally older, less energy efficient dwellings). The results reveal that the inclusion of PPV has substantial bearing not only on the magnitude, but also the direction of health impact. Without PPV large increases occur in exposures to pollutants derived from indoor sources (Table 6), which more than offset the benefits of improved indoor heating season temperatures and protection against outdoor air pollution, resulting in substantial negative consequences for health overall in both settings.

Table 7
Modelled health impacts:changes in life years over 40 years for each scenario and per 1000 population (brackets).

Scenario	Modelled change in life years over 40 years*	
	London	Milton Keynes
With purpose-provided ventilation		
Resilient and low-carbon scenarios	849,800 (108.6)	21,200 (86.4)
Resilient + scenario	856,500 (109.5)	21,400 (87.2)
Without purpose-provided ventilation		
Resilient and low-carbon scenarios	−1,043,900 (−133.4)	−13,800 (−56.2)
Resilient + scenario	−1,041,000 (−133.0)	−13,700 (−55.8)

*The period of the study 2010–2050.

4. Discussion

- There are substantial differences in results for the two locations when housing interventions are the sole mechanism for decarbonization, without additional substantial grid decarbonization. London housing can achieve greater reductions in CO_2 emissions (and possible average health net benefits) than that of Milton Keynes (Summerfield et al., 2007). This is as a result of various factors; the Milton Keynes housing stock is more recent, built to a higher energy efficiency standard. Type and distribution of housing differs appreciably with over 50% of London's stock being purpose built flats (requiring simpler measures to obtain gains), while Milton Keynes stock comprises 80% detached, semi-detached and terraced dwellings with generally larger building volumes that result in smaller concentrations of pollutants per unit volume (MKiO, 2014). To achieve similar reductions in CO_2 emissions in Milton Keynes would require a greater investment in more technical housing interventions such as mechanical ventilated heat recovery (MVHR) systems. The appropriateness of such interventions would of course require a detailed cost-benefit analysis. However, based on our study, it would appear that potential CO_2 reductions and health impacts (whether positive or negative) are stock-specific and policies should be tailored to take this into account rather than be universally rolled out, with both regional and local strategies focussing on the most appropriate sectors in order to achieve CO_2 emissions reduction targets.
- In both London and Milton Keynes, changes to the indoor environment following combined energy efficiency and PPV interventions (if perfectly implemented) would lead to lower CO_2 emissions, reductions in indoor pollutant concentrations, and increases in indoor winter temperatures yielding average net health benefits. In this respect our results are consistent with those of other published research. (Wilkinson et al., 2009; Crump, 2011; Milner et al., 2014). In scenarios where PPV (properly implemented) is used in conjunction with energy efficiency measures, the overall *per capita* health benefits (including all pollutant exposures and temperatures change) are greater in London, with benefits for cardiopulmonary health due to reductions in indoor exposure to both indoor and outdoor-generated $PM_{2.5}$. There would also be substantial reduction in lung cancer burdens due to the reduced $PM_{2.5}$ and radon levels with a minimal impact on ventilation heat loss in both locations. Providing PPV has impact on energy use of +8.6% on average between the different scenarios, whilst potentially yielding substantial health gains. However, a distribution of impacts will occur because of different housing geometries and occupant behaviours and for some homes and behaviours indoor exposures would increase and there would be health dis-benefits for some people. Approved Document F1 of the building regulations states that following retrofitting ventilation should not become worse (HM Government, 2010), however on-site monitoring would suggested this is not always the case (Sinnott and Dyer, 2012).
- The UKERC carbon intensity scenarios used here assume the use of electricity as the energy source for space heating in the

Table 6
Mean indoor pollutant concentrations and temperatures, 2010 and 2050, for each of the scenarios with/without purpose-provided ventilation.

EE and LCS scenarios	London			Milton Keynes		
	Current	With PPV	Without PPV	Current	With PPV	Without PPV
Exposures to	2010	2050	2050	2010	2050	2050
Standardized indoor temperature (SIT, °C)	17.65	18.00	18.18	17.87	18.15	18.33
ETS	1.00	0.84	2.34	1.00	0.88	1.68
Indoor $PM_{2.5}$ derived from outdoor sources (μg/m^3)	5.52	4.00	2.76	5.55	3.80	3.11
Indoor $PM_{2.5}$ derived from indoor sources (μg/m^3)	12.54	5.12	16.11	9.48	4.11	11.81
Radon (Bq/m^3)	13.98	11.76	32.73	28.18	26.35	55.06
Mould (% with mould index >1)	17.24	10.05	25.24	9.17	7.05	12.92
EE + Scenario[a]	**London**			**Milton Keynes**		
Standardized indoor temperature (SIT, °C)	17.65	18.14	18.23	17.87	18.28	18.38
Mould (% with mould index >1)	17.24	9.90	25.00	9.17	6.88	12.84

[a] Impacts for ETS, $PM_{2.5}$ and Radon remain constant.

domestic sector, which is seen as essential under Low-Carbon scenarios for the UK energy system in 2050 because it can be generated from a range of renewable and low-carbon energy sources including nuclear and the use of carbon-capture technologies (UKERC, 2013). By combining housing interventions with decarbonization of the electric grid a substantial contribution to climate goals can be achieved, with targets exceeded in the UKERC Low-Carbon scenario in both locations. However, in both London and Milton Keynes domestic customer fuel consumption is currently 76% gas (DECC, 2014). It is likely that both legislative and incentive means will be needed to promote change from gas to an all-electric grid. If such a change is delayed or does not occur the predicted reductions in CO_2 emissions seen in this study will not be achieved. An energy efficient housing stock with (largely decarbonized) electricity as its fuel represents the upper limit of possible CO_2 savings.

- Modelling analyses such as this study rely on multiple assumptions and many uncertainties. Its results should therefore be interpreted only as indicative and relative rather than as precise calculations of impact. We have provided an section (6) on 'uncertainty in the SCRIBE modelling' in the supplementary data accompanying this publication. uncertainties in the models have also been explored in previous papers by the authors (e.g. Shrubsole et al., 2012; Hamilton et al., 2015). There is currently limited observed data on the impacts of retrofitting strategies on indoor air quality and health to compare against model outputs. Nonetheless, despite these uncertainties, the results provide important indications of likely impacts that can be used to inform policy decisions.

5. Conclusions

This study has investigated the comparative impacts of dwelling-related CO_2 reduction strategies in London and Milton Keynes using integrated housing intervention and energy supply decarbonization scenarios to calculate possible end user energy demand, pollutant exposures and health impacts. Where CO_2 reduction targets are the main policy driver, substantial reductions can be made in London with energy interventions on housing, whereas for Milton Keynes the potential percentage gains are much smaller because of the already more energy efficient housing stock. Potential net benefits or harms for health are also greater in London as measured in terms of *per capita* gains in life expectancy. We highlight the importance of not applying a 'one size fits all' energy saving and CO_2 emission reduction policy, as local differences in housing and the environment may have important bearing on the impacts that can be achieved. Decarbonization of the grid is essential in achieving CO_2 emissions reduction targets, especially in Milton Keynes.

Moreover, when designing for both low energy use and good health, there are important trade-offs between an increase in the airtightness of dwellings and changes in IAQ. If interventions are not correctly applied, there are risks of serious negative health effects. In order to obtain both health gains and promote success in achieving CO_2 emission reduction targets in both locations, policymakers need to consider a wider view that includes strategies to extensively decarbonize the electricity grid with a move away from the reliance on residential use of gas.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.atmosenv.2015.08.074.

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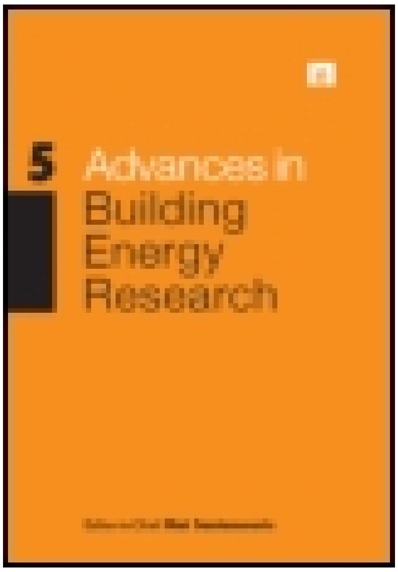

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Impacts of energy efficiency retrofitting measures on indoor $PM_{2.5}$ concentrations across different income groups in England: a modelling study

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As part of an effort to reduce carbon emissions in the UK, policies encouraging the energy-efficient retrofit of domestic properties are being implemented. Typical retrofits, including installation of insulation and double glazing can cause tightening of the building envelope which may affect indoor air quality (IAQ) impacting occupant health. Using the example of $PM_{2.5}$ (an airborne pollutant with known health impacts), this study considers the influence of energy-efficient retrofits on indoor $PM_{2.5}$ concentrations in domestic properties both above and below the low-income threshold (LIT) for a range of tenancies across England. Simulations using EnergyPlus and its integrated Generic Contaminant model are employed to predict indoor $PM_{2.5}$ exposures from both indoor and outdoor sources in building archetypes representative of (i) the existing housing stock and (ii) a retrofitted English housing stock. The exposures of occupants for buildings occupied by groups above and below the LIT are then estimated under current conditions and following retrofits. One-way ANOVA tests were applied to clarify results and investigate differences between the various income and tenure groups. Results indicate that all tenures below the LIT experience greater indoor $PM_{2.5}$ concentrations than those above, suggesting possible social inequalities driven by housing, leading to consequences for health.

Keywords: unintended consequences; low-income housing; low-income threshold; $PM_{2.5}$; retrofit

1. Introduction

The UK Government, in response to the EU Energy Performance of Buildings Directive (EPBD) and motivated by its own greenhouse gas (GHG) emission reduction targets, has begun to implement policies designed to improve the energy efficiency of both new and existing domestic buildings (HM Government, 2010). With existing dwellings predicted to represent 70–80% of the 2050s building stock (Boardman, 2008; Palmer & Cooper, 2011), much of the energy efficiency gains must be obtained through the retrofit of current properties. Using a number of policy mechanisms, the UK government intends these existing dwellings to undergo extensive retrofitting with a range of measures that will increase airtightness, insulation, provide glazing improvements and improve the efficiency of heating systems in order to help meet the UK's own ambitious GHG

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reduction targets (80% reduction of 1990 emissions by 2050) (DECC, 2012). The likelihood of a wide-ranging series of unintended consequences, caused by policy framing and implementation that is narrowly focused on climate change mitigation, has been previously noted (Davies & Oreszczyn, 2012). These unintended consequences may impact building fabric, human health and well-being, the local and wider society and the environment (Shrubsole, Macmillan, Davies, & May, 2014).

One prominent consequence with implications for population health is the change to Indoor Air Quality (IAQ) and personal exposure to airborne pollutants such as particulate matter (PM), the smaller fractions (aerodynamic diameter of 2.5 microns or less –$PM_{2.5}$) of which are particularly harmful to health (COMEAP, 2009). $PM_{2.5}$ is a significant health issue in the UK, with the 2011 fraction of mortality attributable to particulate air pollution estimated to be 5.4% nationwide (based on outdoor $PM_{2.5}$ exposure), representing in excess of 24,000 deaths in 2011 (ONS, 2012; PHE, 2013).

With the UK population spending around 80% of their time indoors, and around half (48–53%) of their time in their own homes (Kornartit, Sokhi, Burton, & Ravindra, 2010), thebuildings and occupant behaviour have the potential to act as significant modifiers on population exposure to pollution from both outdoor and indoor sources (Crump, 2011; Sharpe &Shearer, 2013). $PM_{2.5}$ from external sources, such as emissions from traffic and industry, may infiltrate dwellings, with building location, height, number of exposed façades, orientation to outdoor pollutant sources and meteorology all impacting the amount of $PM_{2.5}$ entering naturally ventilated dwellings (Godish & Spengler, 2004; Patra et al., 2008). In mechanically ventilated dwellings, if systems are correctly installed and maintained, they can influence air change rates and filter pollutants, thereby reducing $PM_{2.5}$ concentrations from both indoor and outdoor sources (Shrubsole et al., 2012).

Indoor sources of $PM_{2.5}$ may include particulates from regular activities such as the burning of fuels, cooking, smoking and cleaning (Klepeis & Nazaroff, 2006; Long, Suh, Catalano, & Koutrakis, 2001), as well as less frequent but high-emission activities such as construction and refurbishment work (Milner, Dimitroulopoulou, & ApSimon, 2005; Weschler, 2009). In multi-dwelling buildings such as apartment complexes, inter-dwelling transfer of pollutants via party wall permeability may also occur (Jones, Das, Chalabi, et al., 2013). Once present inside a dwelling, $PM_{2.5}$ is removed through deposition and exfiltration, and extraction by any mechanical systems. There is also the potential for re-suspension of deposited particulates due to occupant movement and domestic activities (Gehin, Ramalho, & Kirhner, 2008).

Previous studies have indicated that indoor $PM_{2.5}$ concentrations can be higher relative to external levels due to internal sources (Chen & Zhao, 2011), and that increases in indoor $PM_{2.5}$ levels can occur following energy-efficient refurbishment without additional purpose-provided ventilation (Gens, Hurley, Tuomisto, & Friedrich, 2014). Interventions that lead to increased airtightness without compensatory purpose-provided ventilation have been shown to increase exposure to indoor-sourced $PM_{2.5}$ (Shrubsole et al., 2012; Wilkinson et al., 2009).

The type and quality of dwellings inhabited and the practices of the occupants may vary according to socio-economic status and income level, which may then influence pollution exposure. The UK government, the European Union and many other countries define low-income households as those having a household income less than 60% of the national median income that year (DCLG, 2013). Occupants in houses below the low income threshold (LIT) are more likely to live in smaller dwellings such as apartments, which may have lower air change rates than detached, semi-detached or terraced dwellings due to the reduced number of external facades (Taylor, Shrubsole, et al., 2014). Below LIT households may also differ from the overall building stock in terms of building retrofit levels. In addressing the socio-economic and behavioural issues that influence the adoption of energy efficiency measures, Tovar (2012) concludes that households including single adults, those living alone or in cities, lone parents, and tenants in the private sector are the least likely to adopt cavity insulation, loft insulation, and boiler upgrades. However, Hamilton et al. (2014) showed that dwellings with the highest take-up rates of fabric interventions, for example, cavity wall insulation, loft insulation and glazing (the top 20%) are more likely to be found in areas with low income, in part attributable to council-led retrofits in public housing, and national schemes such as Warm Front and energy supplier obligations such as the Energy Efficiency Commitment (Ofgem, 2005; Warm Front, 2004). These findings indicate a potential difference in pollutant exposure between different income and tenure groups and require investigation to clarify the possible impacts on health and to better inform policies aiming to target and improve energy efficiency of the housing stock (HM Government, 2010).

Occupancy and behavioural differences across income groups may also lead to differing levels of exposure to indoor air pollution. In the UK, there is a strong link between smoking and income class, with 35% of unemployed adults smoking, compared to a rate of 19% in the economically active population (ONS, 2007). While smoking may not necessarily always occur inside the home, 59% of daily smokers surveyed allowed smoking in their homes (ONS, 2007). This is likely to be elevated amongst those with mobility issues who are less able to leave their houses. In addition, extractor fans in poor housing may be more likely to remain unrepaired if broken or to underperform, thereby reducing ventilation (EHS, 2012).

Using the English housing stock as an example, this paper examines how the existing housing stock could modify the exposure to $PM_{2.5}$ from indoor and outdoor sources for those in below LIT housing (and the various tenure groups within) and those in above LIT, for both current and full levels of retrofit. Using EnergyPlus, an energy analysis and thermal load simulation program with a multi-zone airflow and contaminant transport analysis component (US-DOE, 2013), simulations were run for the infiltration of outdoor $PM_{2.5}$ into the indoor environment and indoor-sourced $PM_{2.5}$. Simulations included a set of models representing the range of ages and built forms in the current English housing stock and possible fully retrofitted stocks under the different tenancies. The results for each model were weighted according to the frequency of occurrence for each age and built form combination in the different groups studied in order to calculate the differences in total $PM_{2.5}$ exposure between them. Finally, a series of statistical tests were carried out to further clarify the results and test for differences between the different income and tenure groups.

2. Methods

2.1. *Development of representative archetypes*

The 2010–2011 English Housing Survey is a statistically representative survey, comprising ~16,000 EHS dwelling variants (EHS, 2012). Each variant is associated with a weight depending on its incidence in the English housing stock, in addition to a wide range of data describing dwelling characteristics and their inhabitants. A set of 11 archetypes (Figure 1) were constructed with multiple variants representing the range of built forms in the EHS, using archetypes of dwellings from Oikonomou et al.(2012) and the AWESOME project (2013) and assumed to broadly represent the English domestic stock (readers are referred to these papers for full details of their geometries). Where there were built forms with multiple archetypes (e.g. terraced dwellings), the simulation results were averaged across the variants to determine a single value for the built form. The resultant eight built-form bins are then matched to each EHS entry.

2.2. *Dwelling permeability, retrofit level, and operation*

In addition to the built form, permeability (including current and potential retrofit level), occupancy type, and indoor pollution regime were inferred for each entry in the EHS using relevant

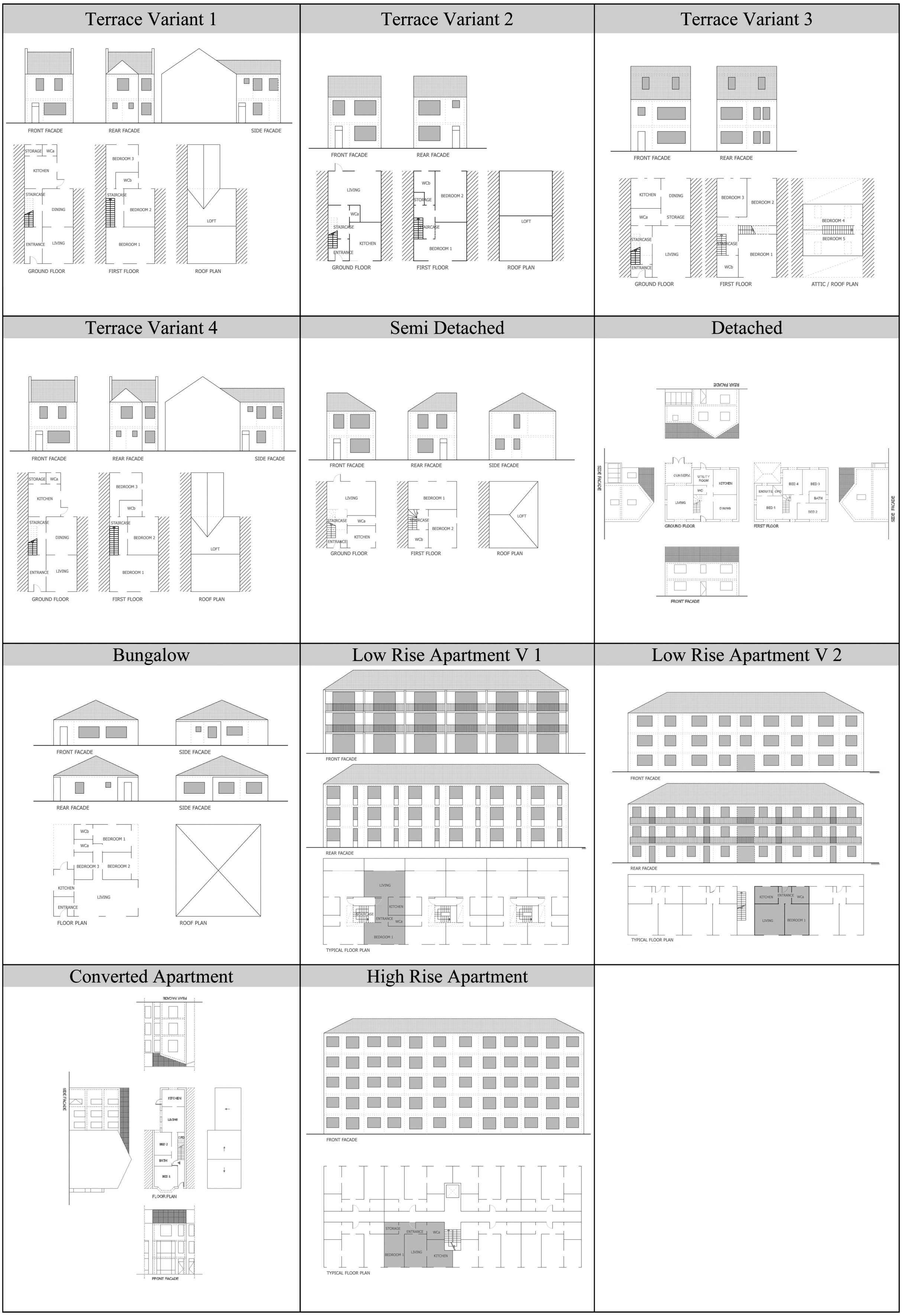

Figure 1. Representative archetypes used to investigate the EHS database.

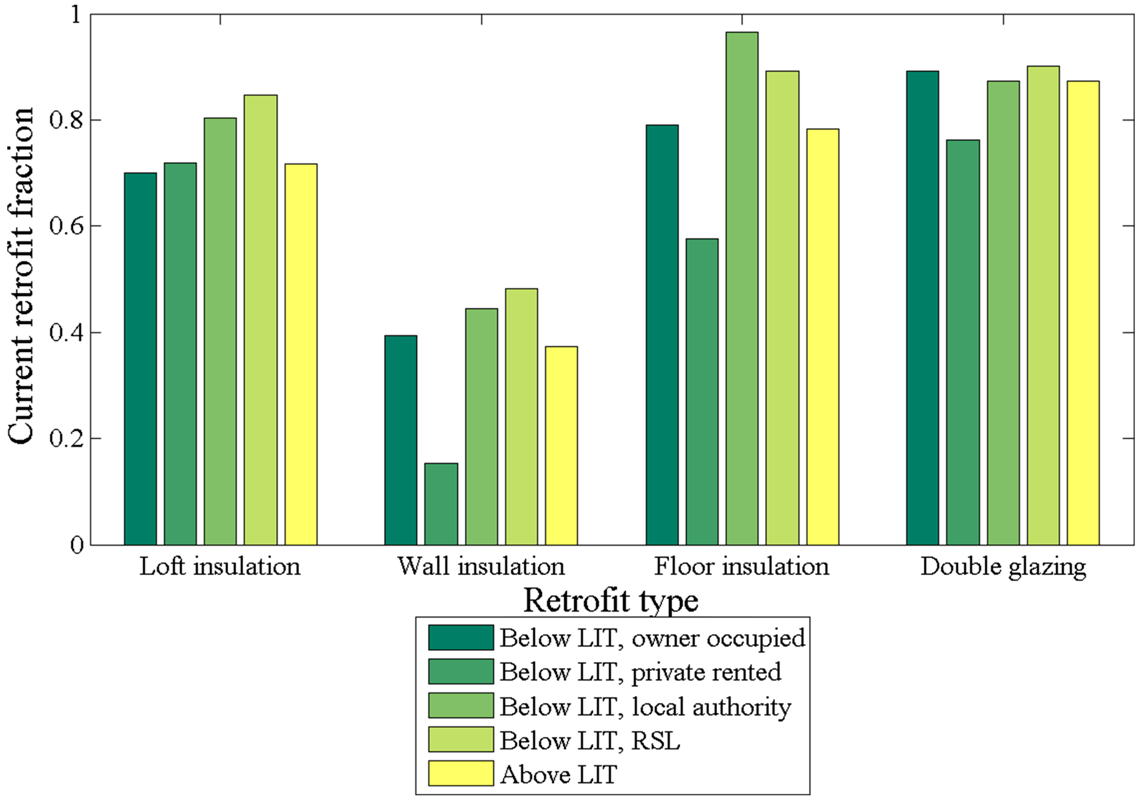

Figure 2. Current levels of various retrofit measures across income and tenure groups.

variables. These variables include: current levels of various retrofit measures, income level after housing costs with respect to the threshold defined in the Introduction, tenure, number of smokers, and the presence of working extract fans. The four potential retrofits examined included wall and loft insulation, floor sealing, and double-glazed windows (used as a proxy for draught-proofing). These retrofits were selected as they are thought to be some of the largest contributors to infiltration according to the Warm Front study (Hong, Ridley, & Oreszczyn, 2004).

Using the EHS data, Figure 2 shows the current levels of retrofit across the various tenure categories within the below LIT group, and for the above LIT-income group. Below LIT private-rented dwellings tend to have the lowest levels of retrofit reflecting the lack of decision-making autonomy for either accepting or seeking energy efficiency improvements. The owner-occupied below LIT and above LIT-income categories have the second lowest levels of retrofit. The below LIT local-authority and registered social landlord (RSL) housing tend to have the highest levels of retrofit (Hamilton et al., 2014). Using the smoking data to determine the presence of at least one smoker in each EHS variant, 44% of below LIT dwellings were found to have at least one occupant who smoked, with similar levels across tenure groups and 28% of above LIT dwellings were found to have at least one occupant who smoked. Analysing the data to determine the presence of working extract fans in the EHS variants found a slight difference in levels of working kitchen extract fans across the income and tenure groups with 44.5% of below LIT-income households and 48.4% of above LIT-income households having a working extract fan.

The permeability of individual dwellings in the EHS was estimated using the UK Standard Assessment Procedure (SAP) methodology (BRE, 2009) as per Taylor, Shrubsole, et al.(2014), with the exception that draught proofing and floor sealing were excluded from the calculation, as their influence on permeability was to be considered separately. Estimated changes to dwelling permeabilities caused by wall, loft, floor, and window retrofits were calculated based on estimates from the Warm Front study (Warm Front, 2004) (Table 1). The current levels of retrofit were

Table 1. Percentage change in permeability following retrofits.

Retrofit measure	Change in permeability (%)
Pre-retrofit (PR)	0
Wall insulation (WR)	−9
Loft insulation (LR)	−14
Floor sealing (FR)	−17
Double-glazing/draught proofing (DGR)	−5

estimated for each dwelling in the EHS, based on the presence of variables reflecting wall, window, and loft improvements, while all pre-1919 dwelling were assumed to have suspended floors and be therefore eligible for floor retrofits (i.e. the sealing or concreting of a suspended floor). The presence of retrofits was used to adjust the SAP-calculated permeability accordingly. Additionally, an estimate of the final permeability following implementation of all four types of retrofit was calculated, providing an estimate of the permeability following a complete building retrofit. It was assumed that retrofits were carried out without any additional compensatory ventilation (a worst-case scenario), and that building permeability did not drop below 3 $m^3/hr/m^2$ based on empirical evidence from currently achieved permeability levels in refurbished and new-build dwellings (Pan, 2010; Sinnott & Dyer, 2012).

2.3. *Dynamic building simulation*

Simulations were constructed and run in EnergyPlus 8.0 using the methodology employed by Taylor, Shrubsole, et al. (2014). Although a short description is provided here, readers are advised to consult this paper for full details. Simulations were run for an entire year with both outdoor and indoor sources of $PM_{2.5}$ (smoking, cooking, and cooking without ventilation). The EnergyPlus (EP) variants comprised each of the built forms modelled at eight different permeability levels (3, 5, 7, 10, 15, 20, 25, and 30 $m^3/hr/m^2$@50 Pa), with the more airtight dwellings (3, 5, and 7 $m^3/hr/m^2$@50 Pa) modelled with fabric characteristics with greater thermal insulation levels. This covered the full range of characteristics of the current and possible fully retrofitted housing stocks under different levels of retrofit. Each EP variant was also modelled assuming four different orientations (North, East, South, and West), to enable orientation-averaged outputs to be evaluated, and both with and without trickle vents. Weather conditions were modelled using a typical reference year weather file for Central London (Islington) obtained from the Prometheus project (Eames, Kershaw, & Coley, 2011) and considered sufficiently indicative of general urban conditions in England for the purposes of this study.

2.3.1. *Occupant behaviour*

A simple single occupancy scenario representative of a family was modelled. The family was assumed to be absent from the dwelling during weekdays between 9am and 5pm, and home all day during the weekends. Dwellings were assumed to be heated to 20°C during the night throughout the year, while internal gains from electrical equipment and occupant metabolism were also included in the model as seen in Taylor, Davies, et al. (2014)

Dwelling window-opening behaviour was coupled to indoor temperatures, as carried out in Taylor, Shrubsole, et al. (2014). Living room windows were considered to be opened during the day if the internal temperatures exceeded 25°C, while bedroom windows were considered to be opened during the night if temperatures exceeded 23°C. In both cases, windows remained closed if the indoor temperatures were less than those outdoors and at times when the dwellings were unoccupied. While there are a number of factors which may influence occupant window-opening behaviour, internal temperature is one of the most significant, and the thresholds used in this study are in line with those observed in field studies (Dubrul, 1988; Fabi, Andersen, Corgnati, & Olesen, 2012) and CIBSE overheating guidelines (CIBSE, 2006).

2.3.2. *Pollutants*

$PM_{2.5}$ levels and emission schedules were modelled as per Shrubsole et al. (2012); the schedule of activities can be seen in Table 2 while the $PM_{2.5}$ emission rates, outdoor particle penetration factor, and deposition rates can be seen in Table 3.

A different deposition rate was considered for Environmental Tobacco Smoke (ETS) due to the different size fraction of $PM_{2.5}$ that characterises the majority of ETS. Two ventilation scenarios were modelled during cooking with the extractor fans either on or off, while no additional ventilation was used when smoking occurred indoors. Although it is likely that the different constituents of $PM_{2.5}$ pose different risks to health, given the lack of evidence in this area, it has been assumed that $PM_{2.5}$ from indoor sources are equally as toxic as those found in outdoor air.

Table 2. Indoor $PM_{2.5}$ production schedules.

Activity	Location	Schedule
Cooking	Kitchen	07:45–08:00
		12:00–12:30[a]
		19:00–19:30
Smoking	Kitchen	8:00–8:05
		9:00–9:05
	Living Room	10:00–10:05[a]
		11:00–11:05[a]
		12:00–12:05[a]
		19:00–19:05
		20:00–20:05
		21:00–21:05
		22:00–22:05

[a]Represents those events that only occur on weekends.

Table 3. $PM_{2.5}$ emission rates, outdoor particle penetration factor, and deposition rates.

Source	Penetration factor	Annual outdoor level	Emission rate	Deposition rate
Outdoor	0.8 when windows closed[a] 1.0 when windows opened[a]	13μg/m^2 [b]	–	0.19 h^{-1} [a]
Cooking	–	–	1.6 mg/min[c]	0.19 h^{-1} [a]
Smoking	–	–	0.9 mg/min[c]	0.10 h^{-1} [d]

[a]Long et al., 2001.
[b]Shrubsole et al., 2012.
[c]Dimitroulopoulou, Ashmore, Hill, Byrne, and Kinnersley, 2006.
[d]Klepeis and Nazaroff, 2006.

2.4. *Data output and analysis*

2.4.1. *Data collation and matching*

The hourly pollutant concentrations in the living room, bedroom, and kitchen were output from the simulations as representing those rooms most frequently occupied. The EP output files were collated and analysed using the SAS statistical package (SAS, 2013), and used to calculate the pollutant concentrations occupants were exposed to, based on the room occupied at the corresponding schedule time. The annual average concentration of $PM_{2.5}$ from outdoor sources (in absolute levels relative to the constant outdoor background of 13 μg/m^3), and from cooking, cooking without extract fans, and smoking (in absolute concentration, μg/m^3) were averaged across the four building orientations for each simulated EP built-form/permeability variant. Exposures were estimated for the primary individual; smoker and/or cook, or people occupying the same rooms during these events.

Indoor $PM_{2.5}$ concentrations from different sources were then assigned to each entry in the EHS based on the built form and estimated current and complete-retrofit permeability by interpolating between the different modelled permeability levels. Dwellings with post-2002 double-glazed windows were assumed to have trickle vents installed, while those installed before were considered to be without trickle vents. The presence of a working extractor fan in the kitchen in each EHS entry was used to indicate whether indoor pollution levels from cooking were with or without such a ventilation system. Smoking was similarly weighted: if a smoker was not present in the EHS variant, the $PM_{2.5}$ concentration from smoking was assumed to be zero. Estimates of the current variation and likely changes in $PM_{2.5}$ exposures following a full retrofit of the housing stock across tenure and income categories were then examined.

3. Results

The mean indoor $PM_{2.5}$ concentrations for the current housing stock and a fully retrofitted housing stock derived from the EP simulations are shown in Figure 3. These include both $PM_{2.5}$ from outdoor sources and indoor sources including smoking and cooking across various income and tenure groups.

The simulations show that cooking is clearly the biggest contributor of $PM_{2.5}$ to the indoor environment and that cooks therefore receive greater exposures than occupants not present in the kitchen. From this, it can be inferred that those who undertake the majority of the household cooking may experience greater levels of exposure compared to non-cooks, whilst they are both exposed to similar levels of externally generated $PM_{2.5}$. There is also a suggestion that below LIT-income groups are at higher risk of exposure to greater concentrations of $PM_{2.5}$ when compared to above LIT-income groups due to smaller houses and a smaller number of exposed facades leading to a reduced air change rate. In addition, they may experience higher rates of smoking and greater likelihood of cooking without working extractor fans. It appears that the fully retrofitted housing stock, with no additional purpose provided ventilation, poses a higher health risk compared to the current housing stock primarily due to a general reduction in building permeability and consequent air change rate following retrofitting interventions on the building envelope. Whilst this has the effect of reducing the ingress of outdoor-sourced $PM_{2.5}$, it results in an increase in concentrations of indoor-sourced $PM_{2.5.}$

A series of one-way ANOVA tests were carried out in MATLAB (MathWorks, 2012) to further clarify the results and test for differences between the income and tenure groups within each of the current and fully retrofitted housing stocks, and between the current and fully retrofitted housing stocks as a whole. As there are more than two groups when comparing between

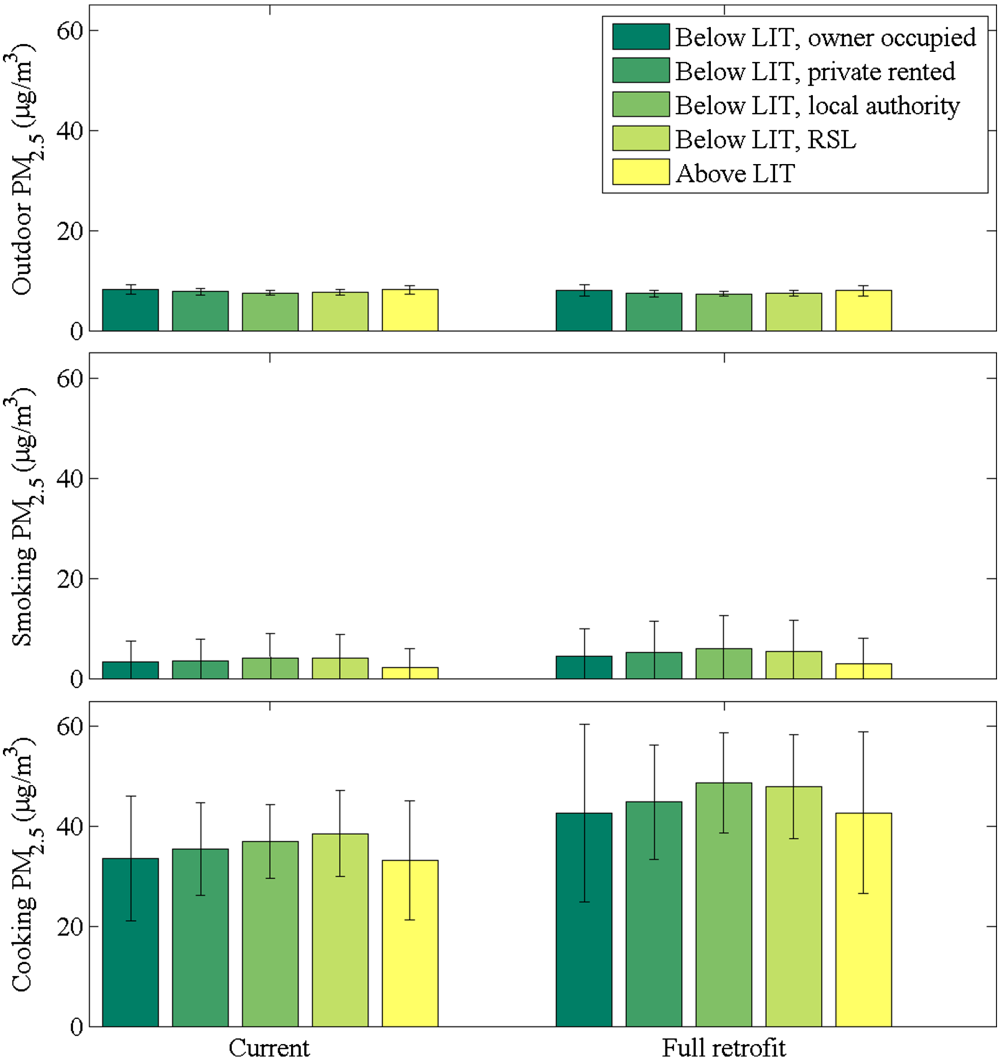

Figure 3. Indoor $PM_{2.5}$ concentrations from different sources for current and fully retrofitted scenarios, and across income and tenure groups. The error bars show standard deviations, and are large for smoking due to some dwellings having zero concentrations. The cooking $PM_{2.5}$ is the exposure experienced by cooks in the kitchen of the properties.

income and tenure groups, MATLAB's multiple-comparison tests were subsequently carried out if the initial ANOVA test found a significant difference. These isolated the location of the differences whilst ensuring that Type-II errors were adequately accounted for. The results are shown in Table 4.

The ANOVA tests support significant differences in all cases at the 95% level of confidence, although it is acknowledged that this reflects differences between the modelled $PM_{2.5}$ exposures rather than actual exposures. Actual exposures may exhibit different distributions as a result of uncertainties in model variables such as the behaviour of occupants, which may also vary across income and tenure groups, dwelling characteristics that are not informed by the described data sources and variations in weather variables across dwelling locations. In the case of comparing modelled exposures between current and fully retrofitted housing stocks, the ANOVA tests highlight significant differences in the concentrations of different sources of $PM_{2.5}$ indoors: outdoor $PM_{2.5}$ decreases, smoking $PM_{2.5}$ increases, and cooking $PM_{2.5}$ increases.

Table 4. Results of ANOVA tests for the different $PM_{2.5}$ sources.

Pollutant	Between current income/ tenure groups	Between retrofitted income/tenure groups	Between current/ retrofitted groups	Details
Outdoor $PM_{2.5}$	Yes	Yes	Yes	Below LIT-income, owner-occupied and above LIT-income groups are similar to each other, but different from other groups in current housing stock, though the groups are more similar in the fully retrofitted housing stock
Smoking $PM_{2.5}$	Yes	Yes	Yes	Above LIT-income group is different from all other groups in both current and fully retrofitted housing stocks
Cooking $PM_{2.5}$	Yes	Yes	Yes	Below LIT-income, owner-occupied and above LIT-income groups are similar to each other, but all the other groups are significantly different from these and from each other in the current housing stock. Similar for fully retrofitted housing stock, but below LIT-income local-authority and RSL groups more similar to each other

Notes: 'Yes' signifies a difference at the 95% level of confidence. The 'details' column summarises the differences as derived from the multiple-comparison tests.

4. Discussion

This study provides new insights into the average relative differences in indoor $PM_{2.5}$ exposure that exist between the various income and tenure groups of the English domestic stock. These differences in exposure are primarily driven by differences in the dwelling characteristics they occupy, but also their habits, such as smoking. In addition, the study describes the potential impacts of changes to occupant $PM_{2.5}$ exposure following an energy efficiency retrofitting scenario. Results generated from the computer modelling were analysed further to determine the statistical limits of the relative differences. Exposure levels modelled are generally consistent with previous research using different modelling programmes and techniques (e.g. Gens et al., 2014; Milner et al., 2005; Shrubsole et al., 2012), which shows that the application of energy efficiency interventions on the domestic stock, whilst reducing exposure to outdoor-sourced $PM_{2.5}$, may increase exposure to indoor sources. The UK government has adopted the various requirements of the EU EPBD; consequently, EU and UK emission reduction goals are similar for the built environment. However, differences in dwelling construction, fuel type (e.g. for cooking) and the provision/or lack of compensatory ventilation across the EU will likely lead to a range of values for exposure to indoor $PM_{2.5}$ (Hanninen et al., 2004).

It is recognised that the choice of occupant schedules and related activities impact the indoor $PM_{2.5}$ exposure. In this paper, a simple schedule was used. The addition of multiple occupant schedules for different income groups would make comparing the relative impact of the building stock and tenures on $PM_{2.5}$ exposure levels more complex. Calculation of exposure to individuals other than the smoker and/or cook, or people occupying the same rooms during these events, is beyond the scope of this paper, but a useful topic for further study. Future work could develop a full range of schedules for different household types and specifically explore occupant behaviour in greater detail.

It would appear that below LIT-income groups have, on average, higher levels of exposure to $PM_{2.5}$ across the building stock when compared to above LIT-income groups. This may in part be due to the greater uptake of measures that reduce the permeability of the building envelop and therefore lower air change rates where additional purpose-provided ventilation is not provided or maintained. However, it is acknowledged that within each income band, there will be a range of individual personal indoor $PM_{2.5}$ exposures. Furthermore, as with all modelling studies, a number of assumptions are required, and further empirical investigation is necessary to confirm or refute the findings. The primary $PM_{2.5}$ source appears to be from cooking, and therefore the provision, use, and appropriate maintenance of adequate extraction equipment (e.g. cooker hoods) are essential to remove this pollutant. This could reduce the apparent increase in $PM_{2.5}$ concentrations and still keep the benefits of increased insulation such as greater thermal efficiency. Assistance with fuel costs, whilst encouraging better ventilation behaviour, may also increase relative CO_2 emissions undermining reduction policies.

Comparisons between groups in each housing stock using the ANOVA multiple-comparison tests show that below LIT-income owner-occupied and above LIT-income groups have higher levels of outdoor $PM_{2.5}$ in the current housing stock, most likely due to the lower levels of retrofit shown in Figure 2. However, these differences are not seen in the housing stock following full retrofit. The above LIT-income groups have lower levels of $PM_{2.5}$ from smoking in both the current and fully retrofitted housing stocks compared to all other groups, primarily as a result of a lower number of households with occupants who smoke rather than other factors. $PM_{2.5}$ sourced from cooking is lower in above LIT-income dwellings in both the current and fully retrofitted housing stocks, and is also lower in owner-occupied and private-rented below LIT-income dwellings compared to local-authority and RSL below LIT-income dwellings. These may be a result of higher levels of retrofit in the local-authority and RSL below LIT-income dwellings, but as these differences persist in the fully retrofitted housing stock, it may also be a result of other factors, for example, generally smaller dwelling/kitchen sizes.

Previous studies have indicated that below LIT-income populations may be exposed to higher levels of outdoor pollution (Pye, Stedman, Adams, & King, 2001; Tonne, Beevers, Armstrong, Kelly, & Wilkinson, 2008), while individuals of low socio-economic groups are the most susceptible to negative health consequences from pollution exposure (Deguen & Zmirou-Navier, 2010). Although in all air-tightening scenarios the ingress of outdoor $PM_{2.5}$ is seen to reduce, it has been demonstrated that in some UK cities (e.g. London), below LIT-income individuals live in areas of higher outdoor $PM_{2.5}$ than the general population (Pye et al., 2001). This may act to counter the advantage of the below LIT-income social housing, which were found to have lower levels of indoor $PM_{2.5}$ from outdoor sources, while further increasing the risks to below LIT-income individuals in privately rented accommodation.

The use of window opening to ventilate dwellings and thereby improve IAQ has been found to be less likely amongst elderly occupants, possibly due to a preference for higher indoor temperatures (Dubrul, 1988; Guerra-Santin, Itard, & Visscher, 2009). No significant correlation was found between socio-economic factors and window-opening behaviour (Dubrul, 1988). However, it is reasonable to assume that in poorer areas where there is either fear of or actual criminal activity, occupants may be less likely to leave their windows open for security reasons (Fabi et al., 2012). Other factors influencing indoor domestic $PM_{2.5}$ exposure in below LIT-income dwellings that require further investigation are the possibility of overcrowding, and multiple smoking occupants which are known to be more prevalent in below LIT-income dwelling and add to the $PM_{2.5}$ exposure risk. In addition, the reductions in permeability which decrease air change rates may encourage the transmission of airborne infections and diseases in below LIT-income properties (Beggs, Noakes, Sleigh, Fletcher, & Siddiqi, 2003; Noakes, Beggs, Sleigh, & Kerr, 2006). This is particularly relevant to the private-rented sector which is currently growing and is less regulated when compared to Local Authority or RSL dwellings.

It is acknowledged that whilst $PM_{2.5}$ has known negative health impacts, there are other indoor airborne pollutants, for example, volatile organic compounds, radon and mould which each have associated health effects (Milner et al., 2014; Wilkinson et al., 2009). The trade-off that exists between airtightness and the consequent reduction of ventilation heat loss to achieve GHG reduction goals and public health concerns for IAQ have been previously noted (Davies & Oreszczyn, 2012; Wilkinson et al., 2009). Consequently, an inclusive optimum strategy approach is needed for building ventilation (Jones, Das, Biddulph, et al., 2013) if health is to be a key driver of policy rather than a singular focus on decarbonisation (Crump, 2011).

This trade-off between the need for adequate ventilation to improve IAQ, comfort and energy conservation on a limited budget may also add to personal $PM_{2.5}$ exposure profiles for below LIT-income occupants. Airtightening in order to conserve energy will likely also have the effect of raising indoor temperatures during summer months (Mavrogianni, Wilkinson, Davies, Biddulph, & Oikonomou, 2012). This may lead to changes in occupant ventilation behaviour influencing IAQ. This dilemma has been successfully investigated in our study by using coupled thermal/pollutant modelling that is able to account for the increase in outdoor-sourced $PM_{2.5}$ found indoors when occupants ventilate their properties when temperatures become uncomfortably high in the summer. It has been also been noted that $PM_{2.5}$ external levels are generally lower in the summer mainly due to metrological impacts, primarily convection and dispersal (McMurry, Shepherd, & Vickery, 2004), thereby lessening this effect; however, this may not be the case for all pollutant types. In addition, the lower I/O ratio seen in below LIT-income housing may be offset by the location of many such properties in areas with generally higher outdoor pollution levels as previously noted.

5. Conclusions

This study has developed and applied a series of stock model simulations in EnergyPlus in order to quantify the changes in indoor domestic $PM_{2.5}$ exposure within the English housing stock that occur when buildings are retrofitted with energy efficiency measures. These results have been further subjected to a rigorous statistical analysis to confirm trends of the differences in model estimates. This study highlights the unintended consequence of changes to indoor domestic $PM_{2.5}$ exposures and the health trade-offs that may occur when policies to mitigate climate change do not take into account wider health outcomes. Although the English housing stock has been used in this study, many of the conclusions can be applied to the European building stock as a whole. Results indicate that, on average, all types of low-income households below the LIT experience greater overall concentrations of $PM_{2.5}$ than those above the LIT and suggest possible social inequalities driven by housing, leading to consequences for health. Below LIT-income properties are generally shown to be more vulnerable to increased levels of indoor $PM_{2.5}$ from indoor sources when compared to above LIT-income properties, with $PM_{2.5}$ from cooking being the main cause. The increased use of extraction equipment at source could remedy this. Below LIT-income housing represents a complex situation with multiple factors – physical, social, and economic – influencing occupant exposure to pollutants such as $PM_{2.5}$. Whilst tightening the building envelope to save energy and assist with climate change mitigation objectives is necessary, it is also essential that adequate purpose-provided ventilation is provided to avoid the negative health impacts.

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Health effects of home energy efficiency interventions in England: a modelling study

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ABSTRACT

Objective: To assess potential public health impacts of changes to indoor air quality and temperature due to energy efficiency retrofits in English dwellings to meet 2030 carbon reduction targets.

Design: Health impact modelling study.

Setting: England.

Participants: English household population.

Intervention: Three retrofit scenarios were modelled: (1) fabric and ventilation retrofits installed assuming building regulations are met; (2) as with scenario (1) but with additional ventilation for homes at risk of poor ventilation; (3) as with scenario (1) but with no additional ventilation to illustrate the potential risk of weak regulations and non-compliance.

Main outcome: Primary outcomes were changes in quality adjusted life years (QALYs) over 50 years from cardiorespiratory diseases, lung cancer, asthma and common mental disorders due to changes in indoor air pollutants, including secondhand tobacco smoke, $PM_{2.5}$ from indoor and outdoor sources, radon, mould, and indoor winter temperatures.

Results: The modelling study estimates showed that scenario (1) resulted in positive effects on net mortality and morbidity of 2241 (95% credible intervals (CI) 2085 to 2397) QALYs per 10 000 persons over 50 years follow-up due to improved temperatures and reduced exposure to indoor pollutants, despite an increase in exposure to outdoor-generated particulate matter with a diameter of 2.5 μm or less ($PM_{2.5}$). Scenario (2) resulted in a negative impact of −728 (95% CI −864 to −592) QALYs per 10 000 persons over 50 years due to an overall increase in indoor pollutant exposures. Scenario (3) resulted in −539 (95% CI −678 to -399) QALYs per 10 000 persons over 50 years follow-up due to an increase in indoor exposures despite the targeting of pollutants.

Conclusions: If properly implemented alongside ventilation, energy efficiency retrofits in housing can improve health by reducing exposure to cold and air pollutants. Maximising the health benefits requires careful understanding of the balance of changes in pollutant exposures, highlighting the importance of ventilation to mitigate the risk of poor indoor air quality.

Strengths and limitations of this study

- The epidemiological evidence about health effects associated with indoor air pollutants and thermal stress is of varying certainty, though more evidence exists for exposure to outdoor pollution and temperature; therefore, only exposures with strong evidence were used.
- This study uses advanced validated building physics models to determine the change in indoor pollutant and thermal exposures related to energy efficiency retrofits.
- The uncertainty in the exposure responses on estimates of health impacts, such as the estimates for cold-related deaths, the toxicity level of particles derived from indoor sources and mental health, could result in a different balance of pollution impact depending on the assumptions made.
- While offering policymakers a support tool to include health as a criterion when developing and assessing home energy efficiency policy, the results presented here should be viewed with a clear understanding of the limitations associated with a modelling study.

INTRODUCTION

By 2030, the UK housing stock will undergo major changes to improve its energy performance,[1] motivated by the need to reduce emissions of greenhouse gases (GHGs), considerations of energy security/cost, and concern about fuel poverty with its presumed link to the UK's large burden of winter/cold-related mortality and morbidity.[2] Housing is responsible for one-quarter of total UK CO_2 emissions[3] and 52% of this is from space heating. Meeting the UK's ambitious energy efficiency targets will require investments to upgrade the energy performance of nearly all dwellings by 2030.[1] These changes to housing energy performance will comprise one of the largest natural experiments in the indoor environment in the coming decades and these are likely to have major impacts on the indoor environment and population health.[4 5] To date, health consequences have received limited examination,[6] though they are increasingly being recognised as an issue by the UK Government.[7]

Properly designed and implemented, actions to improve housing energy performance could have major co-benefits for public health,[4] although there are risks involved and the possibility of poorly designed interventions leading to unintended consequences (figure 1).[8–10] Energy efficiency retrofits that alter the fabric heat loss can also increase the air tightness of the dwelling,[11 12] increasing exposure to indoor-generated pollutants (eg, particulates, mould, radon). Living in cold or inefficient and poorly ventilated homes is linked to a range of health problems.[5 10 13] Retrofits that improve indoor temperatures may have positive impacts on mental health and cardiorespiratory disease,[5] but could have negative impacts on respiratory conditions due to the increased levels of indoor pollutants.[14 15] In the UK, most of our time is spent indoors and the majority of the health impact of more airtight buildings is likely to occur over the long term through low-dose exposure.[16]

While current English building regulations requires that adequate means of ventilation is provided to dwellings,[17] there is a lack of guidance for determining the level of ventilation required to protect health before or following an energy efficiency retrofit.[18] The only guidance that exists relates to the replacement of existing window trickle vents. Ultimately, additional ventilation following a retrofit is left to the discretion of the installer or household. The aim of this study is to illustrate the potential health impact of energy efficiency retrofits under different ventilation settings.

In this paper, we describe the results of a modelling study to quantify changes in exposures in the indoor environment and their associated health consequences attributable to housing energy efficiency retrofits. We do this to characterise possible health-related consequences in need of further scrutiny for the development of national policies and guidance on housing energy efficiency interventions. By doing so, we attempt to gain a better understanding of the trade-offs between risks and benefits for population health.

METHODS

We developed a household-level model to quantify the principal exposure and health pathways outlined in figure 1. The model comprised two parts:

1. A building physics model of English houses that quantifies indoor winter temperatures, exposures to particle pollution, secondhand tobacco smoke (STS), radon, mould growth and energy demand in relation to the energy performance of the dwelling; and
2. A model of the resulting health impacts based on a combination of life table methods and directly modelled changes in disease prevalence.

The two model components make up the Health Impact of Domestic Energy Efficiency Model (HIDEEM;

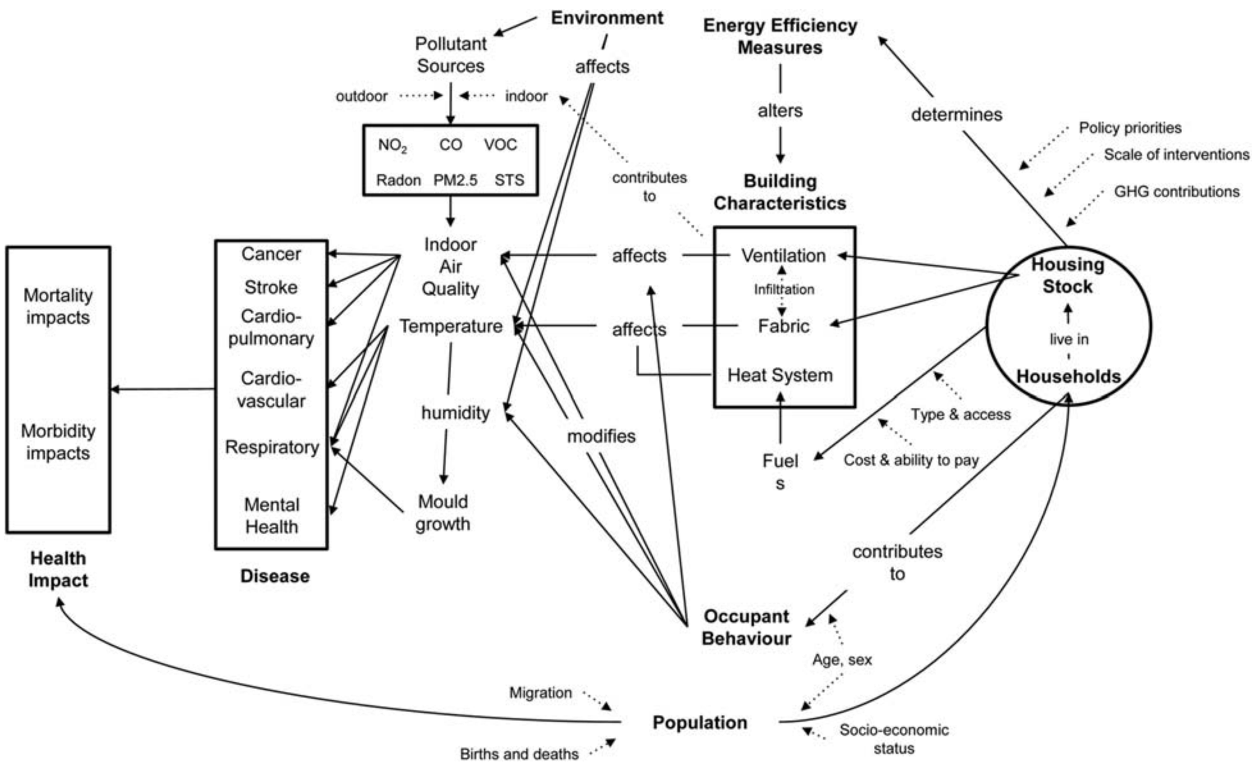

Figure 1 Connections between energy efficiency in housing and health (GHG, greenhouse gas; STS, secondhand tobacco smoke; VOC, volatile organic compound).

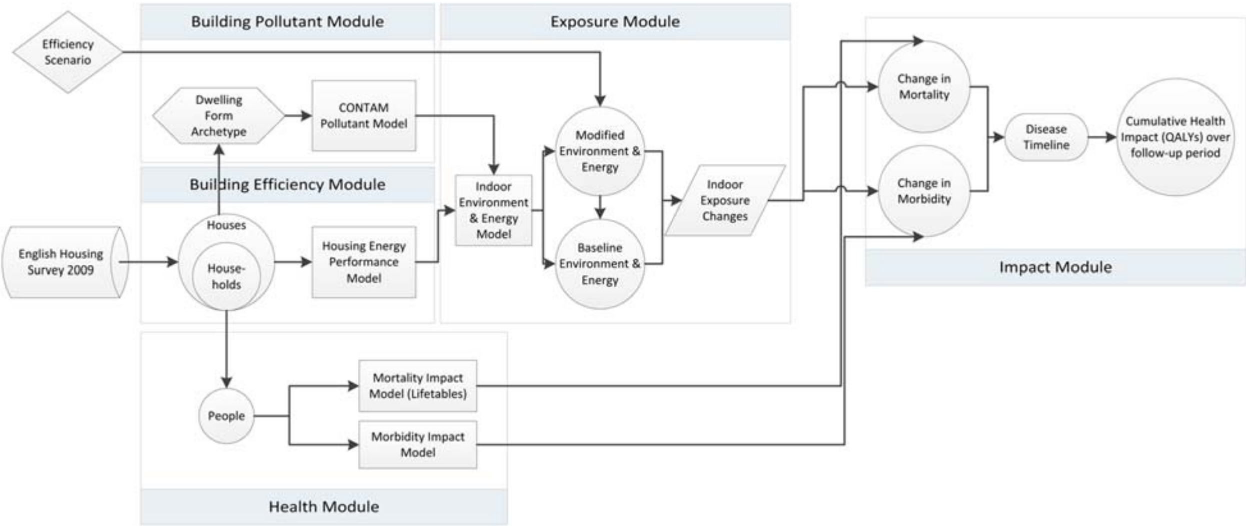

Figure 2 Health Impact of Domestic Energy Efficiency Model (HIDEEM) conceptual framework. The figure demonstrates the components of the model with solid lines representing input flows.

figure 2), an exposure-determinant and health impact model.

Other health outcomes that could be related to energy efficiency interventions but were not considered here include cold-related falls, changes in mental health impact (aside from temperature) and some forms of indoor pollutants (eg, volatile organic compounds, carbon monoxide poisoning, dust mites). However, such evidence can be sparse and the exposure–response uncertain. We have not modelled the impact of cold on respiratory disease (eg, chronic obstructive pulmonary disease) because the evidence required for robust quantification is still equivocal;[19] we hope to address this in future versions of the model. Also, we have not modelled the risk of overheating on energy efficiency, though this could have an important impact in the future. A difficulty with many empirical studies looking at the health effect of energy efficiency interventions is that the study designs and methods have not been sufficiently robust in their design or controlling for bias so as to draw strong conclusions.[5]

Part 1: Modelling the indoor environment

We developed a model that characterised the indoor environmental conditions of the 2010 English Housing Survey (EHS).[20] The indoor environmental conditions and changes in those conditions related to energy efficiency interventions were modelled using validated building physics and airflow models.[21–23] The modelling, described in detail elsewhere,[16 24 25] used representative archetype dwelling forms (informed by sampling from the EHS[26 27]) to represent the English dwelling stock. Each of these archetypes was modelled under different levels of air tightness and ventilation systems: window opening only, window trickle vents, extract fans, and combined use of trickle vents and extract fans. A total of 896 archetypes were modelled and matched to the EHS on the basis of dwelling type (eg, detached, semidetached, terraces and flats), floor area and notional permeability. The result was a model of indoor environmental conditions for a representative sample of English dwellings (see online supplementary appendix 1 for further details).

Dwelling energy performance was calculated as a notional heat loss value.[12] We used an empirical relationship between the dwelling heat loss value and standardised internal temperature (SIT)[i] to predict the bedroom and living room temperature, standardised at an external temperature of 5°C.[12 28] The SIT is a measure of the thermal condition of the dwelling ranked against all other dwellings, and is a function of the dwelling's energy and ventilation performance. The estimated average SIT (derived from an average temperature of the living room and bedroom) for each dwelling reflects the observed distribution shown in Oreszczyn *et al*.[11] The SIT to thermal performance relationship used in the model captures empirical rebound in temperature (eg, reduced heat flow, changes in occupant heating practices and temperature increases).[12] We used EHS data on dwelling fabric characteristics, heating system type and presence of ventilation systems to determine eligibility for energy efficiency upgrades (see online supplementary appendix 2).

Part 2: Quantification of health impact

We focused on a relatively restricted list of exposures that are supported by reasonably clear epidemiological evidence.[5] The health impact of changes in indoor air

[i]The standardised internal temperature (SIT) is derived from an empirical study of 1600 English dwellings with half-hourly temperature measurements for a period of 2–4 weeks over the winter period of 2001/2002 and 2002/2003. The SIT is derived from regression models of indoor on outdoor temperature for each dwelling. The models are used to derive a predicted indoor temperature at 5°C outdoor temperature.[12]

Table 1 Mortality and morbidity outcomes modelled and exposure–response relationships

Exposure	Health outcome	Exposure–response relationship	Reference
Mortality			
Standardised internal temperature	Winter excess cardiovascular (including excess cerebrovascular accident and myocardial infarction)	0.98 per °C	Derived from ref. 32
Secondhand tobacco smoke	Cerebrovascular accident	1.25 (if in same dwelling as smoker)	33
	Myocardial infarction	1.30 (if in same dwelling as smoker)	34
$PM_{2.5}$	Cardiopulmonary	1.082 per 10 μg/m^3	35 36
	Lung cancer	1.059 per 10 μg/m^3	As above
Radon	Lung cancer	1.16 per 100 Bq/m^3	37
Morbidity			
Standardised internal temperature (°C)	Mental health: Common mental disorders (GHQ-12 score 4+)	0.90 per °C	Based on Warm Front[38]
Mould (% MSI >1)	Asthma		
	Harm class II (hospital admission)	1.53 per 100%	Based on ref. 39 and used in HHSRS*
	Harm class III (GP consultation)	1.53 per 100%	As above
	Harm class IV (minor symptoms)	1.83 per 100%	As above

*Housing health and safety rating system.
GHQ, General Health Questionnaire; GP, general practitioner; HHSRS, housing health and safety rating system; MSI, mould severity index; $PM_{2.5}$, particulate matter with a diameter of 2.5 μm or less.

quality and temperature on (cause-specific) mortality was modelled using life table methods based on the IOMLIFET model[29] but applied to individuals in the EHS data based on their age, sex and specific exposure changes. Life tables were set up using 2010 age-specific population and (disease-specific and all-cause) mortality data for England and Wales from the Office for National Statistics (ONS), with separate life tables set up for males and females.[30] We modelled changes in five indoor exposures: SIT, STS, indoor and outdoor sources of particulate matter with a diameter of 2.5 μm or less ($PM_{2.5}$), radon and mould; the selected outcomes are listed in table 1. Impacts on morbidity for these same outcomes were estimated from the mortality estimates by applying age-specific and cause-specific ratios of years of healthy life lost due to disability (YLD) to the overall years of life lost (YLL) derived from WHO Global Burden of Disease data.[31]

Since some of the outcomes are subcategories of others, to avoid double counting we removed deaths in those subcategories from the larger categories. For outcomes affected by more than one exposure, we assumed the relative risks were multiplicative.

We assumed no time lags for cold-related deaths since these would likely to begin to occur within a year. For the other outcomes, a change in exposure would not necessarily lead to an immediate change in mortality in the population. Therefore, we incorporated disease-specific time functions to account for disease onset and cessation lags over time. The time lag functions were based on empirical evidence of the effect of smoking cessation on mortality over time,[40] and plausible assumptions about disease progression over time (see online supplementary appendix 3).

We separately estimated morbidity impacts on common mental disorders (CMDs) in adults and asthma in children using published estimates of the underlying disease prevalence in the population to which exposure-related relative risks were applied based on changes in SIT and mould growth, respectively (table 1). Mental health benefit is assumed to persist over 10 years (ie, exponential decay to zero over 10 years).

Model application: 2030 energy efficiency targets

The model was used to examine the effect of energy efficiency retrofits of the type and order proposed under 2030 GHG mitigation pathways for the English housing sector.[1] Where dwellings were eligible, the retrofits comprised installing double glazing, insulating cavity and solid walls, adding loft insulation, installing new condensing gas boilers, and adding draught proofing to improve dwelling air tightness in leaky dwellings (air leakage rate ≥7 m^3/m^2/h). In addition, non-operational extract fans in the kitchen and bathroom were repaired and window trickle ventilators[ii] were installed with glazing upgrades.

We examined three scenarios that addressed ventilation alongside the energy efficiency retrofits (table 2). They were:

[ii] A small purpose provided opening in a window or building envelope that facilitates ventilation in spaces when large openings (windows and doors) are closed and fans are turned off.

Table 2 Energy efficiency interventions modelled

Experiment energy efficiency retrofits	Ventilation scenarios: Regulation	Installer discretion	No added ventilation
	Number of retrofits installed (1000s)		
Loft insulation	5320	5320	5320
Cavity wall insulation	6560	6560	6560
Solid wall insulation	5700	5700	5700
Double glazing installation	2430	2430	2430
Condensing boiler installation	10 730	10 730	10 730
Gas central heating installation	310	310	310
Draught proofing	3870	3870	3870
Trickle vent and extract fans	15 280	900	0
Extract fan installation only	350	350	0
Extract fan refurbishment	50	50	50
Trickle vent installation only	270	270	0

Note that trickle and extract fans include all new installations, extract fan only already have trickle vents, trickle only already have extract fans.

1. Purpose provided ventilation via extract fans and trickle vents (where not already present) was installed to ensure adequate indoor air quality in line with regulations (Regulation);
2. Purpose provided ventilation was installed (or repaired) only for dwellings that exhibit problems of mould or inadequate ventilation as reported in the EHS (~1.16 million dwellings—see online supplementary appendix 1; Installer Discretion); and
3. No purpose provided ventilation was added except for repairing broken extract fans and trickle vents for double glazing to reflect the lack of guidance surrounding energy efficiency retrofits (No Added Ventilation).

We assumed instantaneous installation for all retrofits in order to illustrate the effect of changes in exposures and associated health effect with all other unrelated conditions held constant. We also assumed that no changes occurred in the underlying health status of the population over time, an assumption which previous work has shown to have only a minor effect on life table calculations.[41]

Uncertainty analysis

We used Monte Carlo simulation to assess parametric uncertainty in the health impact estimates associated with the determinant of the exposure change (ie, the change in heat loss and air tightness due to each intervention), the exposure–response relationships and the utility weights for each health outcome. We report 95% credible interval estimates based on the 2.5th and 97.5th centiles of results generated from 500 model iterations.[42 43] See online supplementary appendix 4 for further details.

We also examined the uncertainty of the model due to two important structural assumptions: (1) the length of life lost in those dying of cold-related causes, and (2) the toxicity of particles derived from indoor sources. For cold, assessing chronic health impacts using exposure–response functions based on time series analyses implies that those who are vulnerable to cold-related risks have the same life expectancy as the population average. This is unlikely to be the case; instead it is likely that the people who die of cold-related events are people who have shorter than average life expectancy (see online supplementary appendix 5 for further discussion). To address this, we have examined the effect of assuming that those vulnerable to cold fall into a 'high-risk' subgroup of the population with elevated underlying cardiovascular risk. We then examined the shortening of remaining life expectancy in such a high-risk group as a function of (1) its size as a proportion of the total population (if overall cardiovascular deaths remain the same), and (2) the elevation of risk (relative risk) in the high-risk group compared with the remainder of the population. For particle toxicity, the epidemiology is dominated by studies of outdoor air pollution. However, it is unclear whether the same toxicity should be assumed for particles derived from indoor sources, whose concentration may rise if air tightness is increased. To account for this uncertainty, we performed calculations with and without the inclusion of the estimated effect of particles derived from indoor sources.

There is also uncertainty in the use of the mould severity index (MSI) used in the EHS that is derived from a visual inspection of the occurrence and extent of mould on windows, walls and ceilings. The potential uncertainty of the MSI measurement beyond the simple Monte Carlo treatment of the uncertainty in mould exposure is not examined here.

RESULTS

Indoor environmental exposure levels

The 2030 energy efficiency interventions resulted in improvements in energy performance, as well as appreciable increases in air tightness. The changes in indoor air pollutant concentrations reflected the ventilation

Table 3 Building performance and indoor environment conditions in the English stock for present day (baseline) and cumulative health effect after 50 years for selected exposure-specific diseases under the 2030 energy efficiency retrofit experiment with ventilation scenarios

	Baseline	Experiment ventilation scenarios		
	Intervention stock	Regulation	Installer discretion	No added ventilation
Sample		N		
Dwellings (1000s)		18 990	17 350	17 320
People (1000s)		44 740	41 130	41 060
Building characteristics		Mean (SD*)		
Fabric heat loss (W/K)	294 (167)	219 (120)	213 (115)	213 (116)
Ventilation heat loss (W/K)	75 (45)	70 (42)	51 (35)	50 (33)
Heat system efficiency (%)	76 (12)	88 (11)	89 (10)	89 (10)
Permeability ($m^3/m^2/h$)	16 (5)	11 (5)	11 (5)	11 (5)
Exposure†		Mean (95% credibility intervals)		
Standardised indoor temperature‡ (°C)	17.8 (0.7)	18.1 (18.1, 18)	18.1 (18.1, 18.1)	18.1 (18.1, 18.1)
STS§	0.5 (0.4)	0.5 (0.5, 0.4)	0.7 (0.7, 0.6)	0.7 (0.7, 0.7)
Indoor¶ $PM_{2.5}$ (μg/m^3)	9.4 (5.4)	4.6 (4.4, 4.2)	10.6 (10.1, 9.6)	11 (10.5, 9.9)
Outdoor $PM_{2.5}$ (μg/m^3)	6.2 (1.7)	6.8 (6.5, 6.2)	5.9 (5.6, 5.3)	5.8 (5.5, 5.2)
Radon (Bq/m^3)	22.9 (14.1)	22.4 (20.3, 20.1)	34.2 (30.7, 30)	35 (31.3, 30.7)
Mould (% with MSI >1)	14.9 (7.5)	12.3 (11.6, 11)	18.5 (17.8, 16.2)	18.8 (18.3, 16.5)
Heating energy (MWh/year)	22.9 (10.4)	16.6 (16.4, 16.3)	15.7 (15.6, 15.4)	15.6 (15.5, 15.4)
Health impact**		Total QALYs per 10 000 persons (95% credibility intervals)††		
Cardiovascular (winter)		119 (106, 131)	69 (57, 81)	65 (53, 77)
Heart attack		312 (287, 336)	−232 (−279, −185)	−271 (−319, −223)
Stroke		306 (282, 330)	−258 (−310, −206)	−296 (−349, −242)
Cardiopulmonary		1268 (1169, 1371)	−44 (−83, −6)	−130 (−166, −96)
Lung cancer		233 (209, 258)	−75 (−93, −57)	−97 (−115, −81)
Common mental disorder		2 (2, 4)	3 (3, 4)	3 (3, 4)
Asthma (children)		1 (4, 7)	−1 (−8, −4)	−1 (−9, −5)
Net impact		2241 (2085, 2397)	−539 (−678, −399)	−728 (−864, −592)

*Standard deviation is given for building characteristics as a measure of spread.
†Weighted average values of kitchen (10%), lounge (45%) and bedroom (45%).
‡Average between living room and bedroom temperature when 5°C outdoors.
§STS 1=average exposure level of smoking household.
¶Indoor sources of $PM_{2.5}$ relate to cooking only with an emission rate of 1.6 μg/min.
**Cardiovascular disease is modelled with equal risk across the population and toxicity of indoor and outdoor $PM_{2.5}$ is considered equal and as such the results are likely overestimating the impact—see uncertainty analysis for tests.
††Credibility intervals are derived from Monte Carlo analysis showing using the 5th and 95th centiles from 1000 model iteration results as limits.
MSI, mould severity index; $PM_{2.5}$, particulate matter with a diameter of 2.5 μm or less; STS, secondhand tobacco smoke; QALYs, quality adjusted life years.

strategy applied under the three different scenarios.[iii] Table 3 summarises the energy performance, indoor environmental conditions, changes in exposure levels and health impacts.

Scenario 1 (Regulation), where ventilation systems were added alongside all fabric and heating retrofits, resulted in a 30% reduction in annual heating energy demand, which is aligned with government objectives.[2] Wintertime temperatures increased by 0.3°C on average (with a SD of ±0.5), while added ventilation reduced indoor sources of pollutants (53% for $PM_{2.5}$, 11% for radon, 13% for STS, 23% for mould), but increased indoor exposure to outdoor-generated $PM_{2.5}$ (4.2%).

The 'Installer Discretion' scenario shows that mitigation measures applied due to perceptible conditions of inadequate ventilation or mould growth were insufficient to have wide benefit (in part due to the relatively small number of dwellings exhibiting these conditions, see online supplementary appendix 1). With the added ventilation, heat losses (33%) and heating energy (32%) were greater compared with the 'Regulation' scenario along with a modest increase in indoor temperatures. Outdoor sources of $PM_{2.5}$ reduced considerably (-10%), but indoor pollutants experienced sizable increases (8% for $PM_{2.5}$, 34% for radon, 33% for STS and 18% for mould).

Under the 'No Added Ventilation' scenario, there were still greater reductions in ventilation heat losses. The average indoor pollutant concentrations were

[iii]The modelled estimates for the baseline housing stock energy performance and indoor exposures were compared against observed national and sample stock distributions to check the accuracy of the outputs (see online supplementary appendix 1).

further elevated across the stock compared with scenario 2 (Installer Discretion).

Health impact of energy efficiency retrofits

The balance of the overall impact on mortality and morbidity is highly dependent on the assumptions made regarding the level of ventilation to mitigate reduced indoor air quality (table 3; figure 3). Over a follow-up period of 50 years, the net impact of the 2030 energy efficiency interventions under the 'Regulation' ventilation scenario resulted in 2241 quality adjusted life years (QALYs) gained per 10 000 persons for the 18.99 million affected dwellings. Selective targeting of ventilation system under the 'Installer Discretion' scenario resulted in −539 QALYs per 10 000 persons lost. While no added ventilation had an even greater overall negative impact of −728 QALYs per 10 000 persons lost among the intervention group.

If building regulations were met (scenario 1), the net impact on health is positive primarily because the reduction in exposure to particles of indoor origin is greater than the increase in outdoor-generated particles. Improved indoor temperatures have a net positive effect on cardiovascular disease, though this is dependent on assumptions of the remaining life expectancy of those vulnerable to the effects of cold (see Uncertainty analysis section).

Targeted extract fans and trickle vents in dwellings with a perceptive ventilation problem (scenario 2) offer only moderate modification on the long-term impact on health, a 30% improvement from no additional ventilation (scenario 3). However, despite these interventions, there remained a large number of dwellings that experienced an increase in fabric air tightness.

When no additional ventilation was provided alongside the dwelling energy efficiency retrofits, the increase in indoor sources of air pollutants resulted in a net negative impact on health, despite the reduced ingress of outdoor sources of particulates. Although sensitive to assumptions on the equal toxicity of indoor and outdoor $PM_{2.5}$ (see Uncertainty analysis section), reduced infiltration of outdoor air and increases in exposure to STS, radon and mould risk resulted in a net-negative impact on health.

Uncertainty analysis

Cold-related deaths risk group size

We use here scenario 2 to illustrate the sensitivity of the health impact estimates to changes in the concentration of cardiovascular risk within the population. Reducing the size of the 'high-risk' cardiovascular group in the population reduces the scale of the health benefit due to increased winter temperatures, though the overall impact is modest (see table 4). We illustrate this by concentrating the risk across increasingly smaller proportions of the population (from 100% to 0.1%), selected to represent the full range of plausible assumptions. An assumption of 100% of the excess winter cardiovascular deaths being in the high-risk group (ie, the whole population at risk) could result in a considerable overestimate of the change in the burden of winter time cardiovascular disease, while an estimate of 0.1% (ie, only 0.1% of the population are at risk) would effectively remove all of the potential benefit of increased temperatures for population health. Pending further research, it is difficult to estimate the correct level of adjustment. However, the impact is almost certain to be appreciably less than that implied by using time series coefficients applied without any correction.

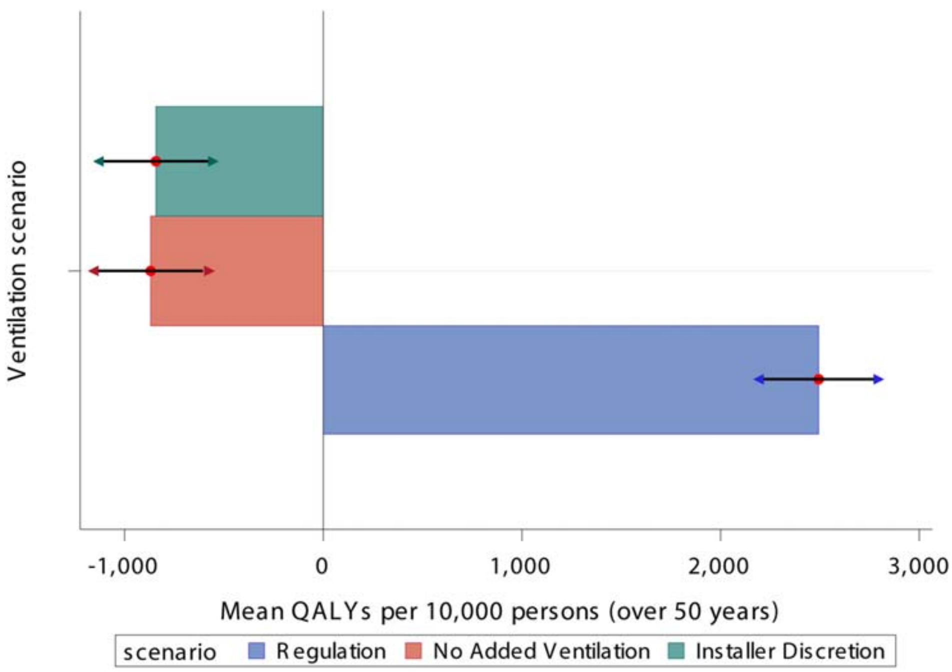

Figure 3 Net mortality and morbidity health effect (quality adjusted life years (QALYs) per 10 000 persons) for all selected exposure-specific diseases after 50 years for the 2030 energy efficiency experiment for different ventilation scenarios (arrows denote 95% credibility intervals). Note: cardiovascular disease is modelled with equal risk across the population and toxicity of indoor and outdoor $PM_{2.5}$ is considered equal and as such the results are likely overestimating the impact—see 'section, Uncertainty analysis' for tests ($PM_{2.5}$, particulate matter with a diameter of 2.5 μm or less).

Table 4 Cumulative health effect after 50 years for varying high-risk excess winter cardiovascular group size under the 2030 energy efficiency retrofit experiment for scenario 2 'installer discretion'

	Experiment ventilation scenario 2: 'Installer Discretion'			
	Size of 'high-risk' group*			
	100%†	10%	1%	0.1%
Net QALYs	**Mean per 10 000 persons (95% credibility intervals)**			
Cardiovascular (winter)	68.8 (56.8, 80.7)	34.1 (28.1, 40)	14.5 (12, 17)	4.8 (4, 5.7)
Heart attack	−232.1 (−279.1, −185.2)	−232.6 (−277.1, −188.1)	−232.7 (−276, −189.5)	−232.2 (−275.3, −189)
Stroke	−257.6 (−309.7, −205.5)	−257.2 (−307, −207.4)	−256.3 (−304.4, −208.2)	−257.3 (−305.5, −209.1)
Cardiopulmonary	−44.3 (−83.4, −5.6)	−46.6 (−85.6, −8.1)	−47.4 (−86.7, −8.8)	−44.2 (−83.4, −5.4)
Lung cancer	−74.9 (−92.9, −57.4)	−74.3 (−91.9, −57.2)	−75 (−92.9, −57.7)	−74.9 (−92.9, −57.5)
Common mental disorder	2.7 (2.8, 4.1)	2.7 (2.8, 4)	2.8 (2.8, 4.1)	2.7 (2.8, 4)
Asthma (children)	−1.3 (−8.4, −4.3)	−1.3 (−8.4, −4.4)	−1.3 (−8.4, −4.3)	−1.3 (−8.2, −4.2)
Net impact	−538.6 (−677.9, −399.3)	−575.2 (−706.5, −443.9)	−595.5 (−724.2, −466.7)	−602.2 (−729.6, −474.8)

*Proportion of the population in the group assumed to be at high risk for cardiovascular events.
†100% equivalent to whole population equally at risk.
QALYs, quality adjusted life years.

Toxicity of indoor particulate matter

There is uncertainty about the relative toxicity of particles generated from indoor sources compared with those from outdoor sources. Some evidence suggests these might be as toxic or perhaps even more toxic as particulate matter (PM) derived from outdoor sources.[35 36] Analysis in which indoor-generated $PM_{2.5}$ was assumed to have no adverse effect on health had a significant impact on the results (see table 5), reducing the overall net health impact by around 78% compared with the base case results (which assumed equal toxicity to outdoor particulates). Though the effect may be uncertain, there is very likely to be some impact from indoor sources and we would stress the need for more empirical studies that measure and assess the toxicity of indoor $PM_{2.5}$, and the balance of indoor and outdoor particles on health.

DISCUSSION

This modelling work shows that predicted changes in indoor environmental exposures following housing energy efficiency interventions of the type being proposed by the UK Government may have an appreciable impact on health. This approach can be applied to different country settings but with regard to existing conditions, and information on the housing stock and households therein.

There is an expectation that retrofits that seek to reduce space heating energy demand will increase indoor temperatures,[12] but such interventions will also affect the dwelling air tightness and its ventilation. Although indicative, our modelling suggests that reducing fabric heat loss and increasing air tightness may reduce exposure to outdoor pollutants and raise indoor temperatures. However, without added ventilation, indoor concentrations are increased with associated adverse health impacts which are greater than those associated with indoor temperatures, leading to an overall negative impact on health. As demonstrated, this conclusion is sensitive to assumptions made about the toxicity of particles from indoor sources, an area where further research is urgently needed.

In the various scenarios, for purposes of illustration, we assumed an instantaneous installation and a lagged health impact associated with step changes in some exposures. However, the reality will be that these interventions and potential impacts will be realised over a longer period of time. Under the UK's mitigation targets, virtually all English dwellings will need retrofitting by 2030 (ie, 20 million over 15 years or 3650 per day). Putting in place effective measures to address ventilation now can have long-term health effects for both existing and future households.

Although associations between indoor temperatures and mental well-being have been reported,[38] it is unclear how long the benefit to mental well-being would persist following improved temperatures. Given the high

Table 5 Cumulative health effect after 50 years for indoor $PM_{2.5}$ toxicity equal to outdoor sources and with no effect of indoor $PM_{2.5}$ under the 2030 energy efficiency retrofit experiment for scenario 2 'installer discretion'

	Experiment ventilation scenario 2	
	Indoor particulate matter toxicity	
	Equal to outdoor	No effect
Net QALYs	Mean per 10 000 persons (95% credibility intervals)	
Cardiovascular (winter)	68.8 (56.8, 80.7)	81.6 (69.8, 93.4)
Heart attack	−232.1 (−279.1, −185.2)	−186 (−225, −147)
Stroke	−257.6 (−309.7, −205.5)	−212.1 (−255.1, −169)
Cardiopulmonary	−44.3 (−83.4, −5.6)	200.8 (170.5, 233.5)
Lung cancer	−74.9 (−92.9, −57.4)	−47 (−59.8, −34.5)
Common mental disorder	2.7 (2.8, 4.1)	2.8 (2.9, 4.1)
Asthma (children)	−1.3 (−8.4, −4.3)	−1.3 (−8.1, −4.2)
Net impact	−538.6 (−677.9, −399.3)	−161.2 (−240.3, −82)

$PM_{2.5}$, particulate matter with a diameter of 2.5 μm or less; QALYs, quality adjusted life years.

prevalence of CMD in the population, any small shift can be highly influential on the results. While there is very likely to be benefit that accrues beyond a single year and maybe a seasonal effect for a period afterwards, the long-term benefit will likely be affected by the risk of reoccurring episodes of mental health driven by factors other than thermal environment.

The underlying assumptions regarding housing air tightness and occupant ventilation practices (eg, window opening behaviour) are both extremely important. The EHS shows that 71% of homes have no extract fans (or working extract fans); in other words, these homes are naturally ventilated and thus, the exposure to indoor-generated pollutants will be highly determined by the air tightness of the dwelling and the practices of the occupants. Our model has examined the uncertainty of these practices on our estimates and therefore, provides a reasonable spread on the likely true impact.[43] From our scenarios, we found that added ventilation accompanying efficiency retrofits mitigated the health risk associated with increased air tightness (scenario 1), but that this mitigation must be applied beyond 'problem homes' (scenario 2), only the widespread installation of ventilation systems results in a net benefit to health (scenario 1), and providing no additional ventilation poses a potential risk to health (scenario 3).

The provision of added ventilation to offset potential increases in indoor concentrations of pollutants following fabric energy retrofits is an important issue for public health. While the spirit of the building regulations suggests that adequate ventilation should be provided following changes to a dwelling, there is no explicit guidance for installers on what and when to install such systems. The Housing Health and Safety Rating System provides an 'after-the-fact' route through which remediation of poor indoor air quality could be addressed, but it is both unlikely and undesirable to rely on this system to address issues that could otherwise be avoided. Clearly assumptions on how a household ventilates their dwelling will have an important impact on creating a healthy indoor environment. Dwellings with higher ventilation rates have been shown to have reduced health burdens,[10 44] though the association with air change rates and specific diseases can be equivocal.[45] Occupant ventilation practices have also been shown to be counter-productive to creating a healthy indoor environment. A study of Dutch households showed that many neglect the annual maintenance required to ensure that ventilation system operation is not compromised.[46] Education around ventilation will be essential to minimise exposure to indoor pollutants following retrofits. Our work highlights that the potential health impacts following efficiency retrofits are not necessarily positive and that there may be risk trade-offs that will depend on the retrofit installation regulatory framework. Having stronger regulation around energy efficiency retrofits and ventilation will help to realise multiple benefits (eg, energy savings and health).

STRENGTHS AND LIMITATIONS

Modelling studies provide a method of examining complex problems by drawing together data from a range of sources in order to explore the potential impact of interventions on population health. While quantifying the potential health impact of policy options is preferable over qualitative assessment, doing so is subject to several difficulties, primarily the availability of evidence[47] and the potential to add scientific credibility to uncertain predictions.[48] The modelling also involves many uncertainties. For instance, the limited set of observed data on how such retrofits affect indoor air quality remains an impediment, with only a few studies looking at the determinants of indoor air quality (eg, infiltration).[5] There is a paucity of evidence relating to some of the most important health outcomes—especially in relation to cold.[49] In the overall balance of health calculations, morbidity impacts are potentially larger than those of mortality, for example, the effect of improved temperatures on CMD,[5] but the evidence is still uncertain, and this gap in the research evidence should be addressed.

The modelling results are presented as QALYs; however, it is clear that these changes in disease outcomes would have an impact on health and social care services beyond these utility estimates. As the average age of the UK population increases so too does the demand on health services. Preventative actions, such as improving energy and ventilation performance, may help to mitigate some of this demand.

The exposure modelling in this experiment concentrated on indoor conditions. The experiment did not alter outdoor pollutant concentrations related to proposed energy supply decarbonisation,[1] which may reduce outdoor levels of particulate matter in the future.[50] This would further tip the balance towards installing mitigating ventilation systems so as to dilute 'stale' indoor air. Refining the model to include assumptions on energy systems and transport could further improve the estimates of the potential health impact associated with UK's GHG abatement measures.

CONCLUSIONS AND POLICY IMPLICATIONS

On balance, if properly implemented, actions to mitigate climate change through energy efficiency in housing can have benefits to health by reducing exposure to cold and outdoor air pollutants. They will also offer indirect health benefits by providing more resilience to protect indoor thermal conditions during extreme cold and heat events. Modelling studies of the type presented here are needed to ensure housing policies are developed in ways that capitalise on this potential for improving health. Such studies, however, should be used with acknowledgment of their uncertainty and limitations, and do not supplant the need for well-designed empirical studies that can validate models and offer policymakers more evidence, and provide greater confidence around policy impact.

We have shown that, unless specific remediation is used, reducing the ventilation of dwellings will improve energy efficiency at the expense of increased exposure to indoor air pollutants and risk to health. However, an important conclusion of this work is that, with careful attention to retrofit installation and ventilation practices, these potential negative impacts can be removed.

The policy agenda and evidence base on the health impact of home energy efficiency is still evolving. Guidance for installers regarding adequate levels of ventilation to protect health is now needed before the large-scale introduction of energy efficiency measures into the housing stock.

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Contributors IH developed the integrated health and exposure model, and was responsible for developing the experiment and crafting the text. JM and ZC developed the health model and contributed to the text. PD, BJ and CS all developed portions of the exposure models and contributed to the text. MD and PW were project leads, guided the study design and contributed to the text.

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